Preliminary Evaluation of Injuries at the LCP Site in Brunswick, Georgia

Final

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Acronyms

PCBs	polychlorinated biphenyls
PAHs	polycyclic aromatic hydrocarbons
LCP	Linden Chemicals and Plastics
DOI	U.S. Department of the Interior
NRDAs	natural resource damage assessments
NOAA	National Oceanic and Atmospheric Administration
DNR	Georgia Department of Natural Resources

1. Introduction

This document presents a compilation and evaluation of data related to potential natural resource injuries that have resulted from releases of polychlorinated biphenyls (PCBs), mercury, lead, and polycyclic aromatic hydrocarbons (PAHs) from the Linden Chemicals and Plastics (LCP) Superfund site in Brunswick, Georgia. This evaluation is conducted consistent with the U.S. Department of the Interior (DOI) regulations for conducting natural resource damage assessments (NRDAs) at 43 CFR Part 11. The report is organized as follows:

- Chapter 2 describes the site and its resources, the site's history, and the relevant data that are available for the site.
- Chapter 3 describes data relevant to determining the extent to which releases from the LCP site have resulted in natural resource exposure to PCBs and mercury.
- Chapter 4 presents an evaluation of exceedences of surface water and sediment criteria and standards.
- Chapter 5 presents a preliminary evaluation of adverse effect injuries to biological resources, including benthic invertebrates, fish, and birds.
- Chapter 6 presents a preliminary evaluation of injuries to fishery resources resulting from exceedences of consumption advisory thresholds.

The purpose of this document is to assist the natural resource trustees [the National Oceanic and Atmospheric Administration (NOAA), the Georgia Department of Natural Resources (DNR), and the United States department of the Interior through the United States. Fish and Wildlife Service] in their evaluation of data that are relevant to an assessment of injuries resulting from LCP site hazardous substance releases. This document does not constitute a finding that natural resource injury has occurred, nor is it a formal injury determination and quantification. No new data were collected for this report, and this reported is based primarily on information and data that were available as of August 2000. The analyses presented in this report may be updated as new information becomes available.

2. Site Description and Available Data

2.1 Site Description

The LCP Superfund site is located just northwest of Brunswick, Georgia, in St. Simons Sound (Figure 2.1). The site includes, but may not be limited to, an approximately 80-acre industrialized upland portion and an approximately 550-acre undeveloped salt marsh (U.S. Fish and Wildlife Service, 2000), where the primary vegetation is marsh grass (*Spartina alterniflora*) (Sprenger et al., 1997). Most of the marsh is inundated at high tide (GeoSyntec Consultants, 1999). Several small watercourses run through the marsh, including the LCP ditch that begins at an outfall near the upland portion of the site. The marsh drains into Purvis Creek, a tidally influenced saltwater creek with a tidal range of 6 to 7 feet. Purvis Creek in turn flows into the Turtle River, approximately three-quarters of a mile downstream of the site (Sprenger et al., 1997; PTI and CDR, 1998).

The area provides habitat suitable for a variety of organisms; Table 2.1 shows that many fish, shellfish, birds, reptile, mammal, and other invertebrate species have been documented in or near the Purvis Creek marsh. The information in Table 2.1 comes from site-specific data reported in the ecological risk assessments (ERAs) performed by Sprenger et al. (1997) and PTI and CDR (1998), and from Odom (1975), Georgia DNR (1996), and Kannan et al. (1998).

Several of the species that have been observed in the Purvis Creek marsh are federally listed species. Wood storks, a federally listed endangered species, have been observed foraging in the marsh and breed at several colonies in the Brunswick area (PTI and CDR, 1998). The West Indian manatee, a federally listed endangered species, has been observed in Purvis Creek and the Turtle River (PTI and CDR, 1998). In addition, several federally listed threatened or endangered species inhabit St. Simons Sound but have not been specifically observed in the Purvis Creek marsh. These species include the shortnose sturgeon, the green turtle, Kemp's ridley turtle, hawksbill turtle, loggerhead turtle, leatherback turtle, and bald eagle (PTI and CDR, 1998).

2.2 Site History

Numerous industrial operations have occupied the LCP site. The Atlantic Refining Company first used the site as a petroleum refinery, from 1920 through 1955. The transfer, processing, and storage of petroleum products during this period resulted in PAH and lead releases at the site (U.S. Fish and Wildlife Service, 2000). PAHs were also suspected to have been released between 1937 and 1950 by Georgia Power, which operated a power generating facility on part of the site (U.S. Fish and Wildlife Service, 2000). Dixie Paint and Varnish Company operated a paint and varnish manufacturing facility on a 10.5-acre portion of the site between 1941 and 1955, and

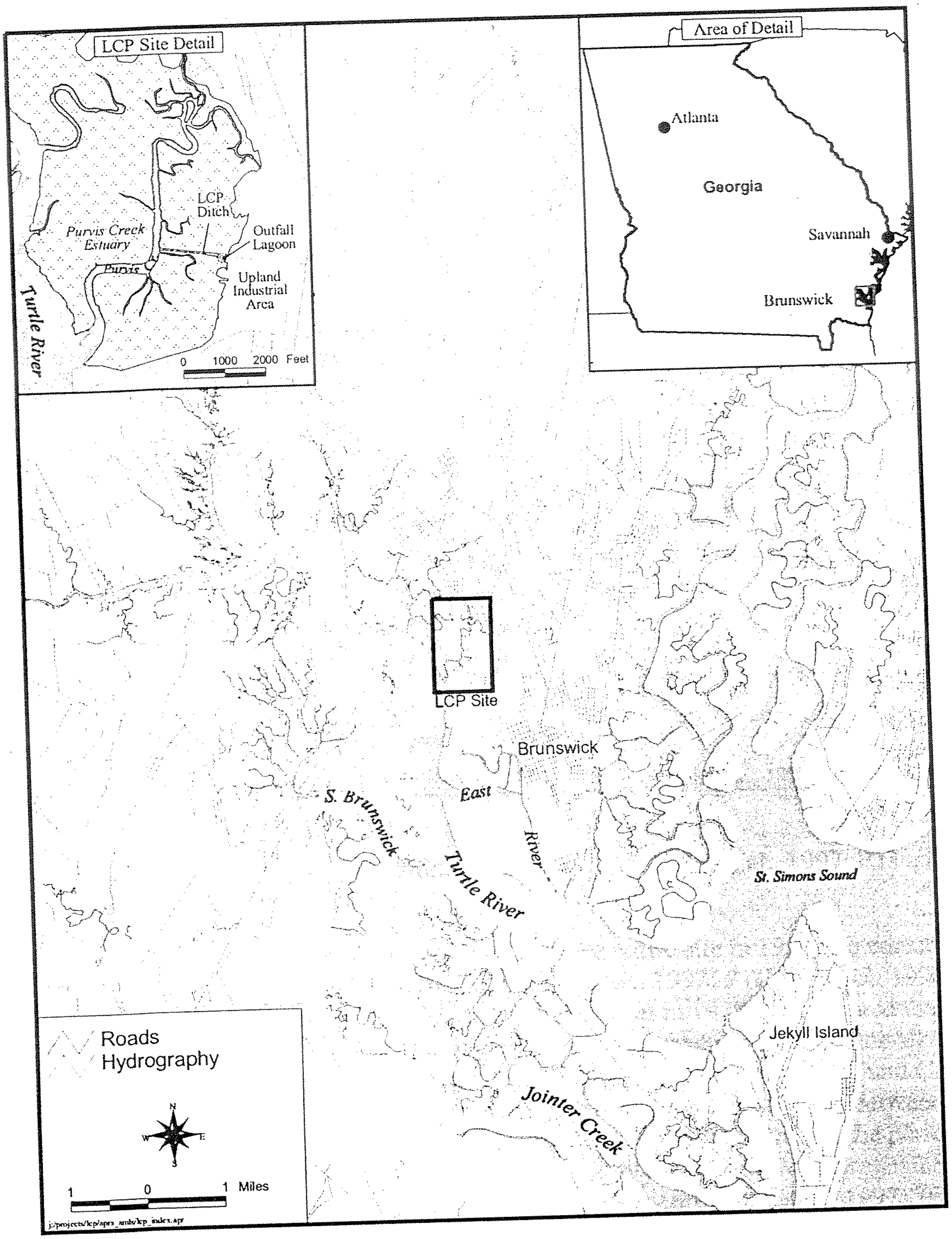

Figure 2.1. The location of the LCP site.

Table 2.1. Selected species common to Georgia saltmarshes and/or documented in the Purvis Creek marsh.

Fish	Shellfish	Birds	Reptiles	Mammals	Other invertebrates
Black drum	American oyster	Boat-tailed grackle	Diamond-back terrapin	Atlantic bottle-nosed dolphin	Periwinkle snail
Blue fish	Blue crab	Clapper rail		Bats	Crustaceans
Croaker	Carolina marsh clam	Great egret		Cotton rat	Gastropods
Flounder	Grass shrimp	Least tern		Marsh rabbit	Insects
Killifish	Fiddler crab	Louisiana heron		Marsh rice rat	Nematodes
Mullet	Brown shrimp	Marsh hawk		Mink	Oligochaetes
Red drum		Marsh wren		Raccoon	Polychaetes
Seatrout		Mottled duck		River otter	
Sheepshead		Red-winged blackbird		Shrews	
Sheepshead minnow		Snowy egret		West Indian manatee[a]	
Spot		Sora			
Spotted seatrout		White ibis			
Striped mullet		Wood stork[a]			
Summer flounder					
Whiting					
Yellow tail					

a. Federally listed rare, threatened, or endangered species.

Sources: Odom, 1975; Sprenger et al., 1997; Georgia DNR, 1996; Kannan et al., 1998; PTI and CDR, 1998.

most likely released PAHs and metals during this period (Sprenger et al., 1997; PTI and CDR, 1998; U.S. Fish and Wildlife Service, 2000).

Allied Chemical and Dye Company purchased nearly the entire site in 1955 and constructed a chlor-alkali chemical manufacturing facility (Sprenger et al., 1997; PTI and CDR, 1998). LCP Chemical-Georgia, Inc. purchased the site from Allied Signal in 1979 and operated the chlor-alkali facility until 1994 (Sprenger et al., 1997). The primary products of the chlor-alkali operation were chlorine gas, hydrogen gas, and sodium hydroxide solution (PTI and CDR, 1998). The chlor-alkali process involved passing a concentrated brine solution between a stationary graphite or metal anode and a flowing mercury cathode. For part of the period in which the site was used as a chlor-alkali facility, the graphite anodes were impregnated with Aroclor 1268, a highly chlorinated commercial PCB mixture (Sprenger et al., 1997). This is the only known use of Aroclor 1268 at the site.

The mercury and PCB wastes found on the site are attributable to the operation of the chlor-alkali plant by Allied and LCP. Allied and LCP released mercury into Purvis Creek continually between 1955 and 1994 (U.S. Fish and Wildlife Service, 2000). Allied built disposal ponds along a tributary to Purvis Creek using PCB-contaminated anodes from the chlor-alkali plant as filler material in the berms. Mercury-contaminated sludge wastes were disposed of in the unlined ponds, which were breached on several occasions. Breaching of pipelines along the south edge of the property, one of which carried sodium hypochlorite, was another source of contamination for the Purvis Creek marsh. It is estimated that more than 440 tons of mercury and 37 tons of PCBs were released at the site (U.S. Fish and Wildlife Service, 2000).

Based on the ERA conducted by the U.S. Environmental Protection Agency (U.S. EPA) Environmental Response Team (Sprenger et al., 1997), the U.S. EPA determined that a removal action in the marsh was warranted (PTI and CDR, 1998). The removal action, which took place between January 1998 and July 1999, focused on marsh areas with high levels of PCBs and mercury known at the time (GeoSyntec Consultants, 1999). Contaminated soils and sediments were removed from areas adjacent to the outfall pond and bordering the upland industrial site, and from two tidal channels, the LCP ditch, and a natural drainage channel that intersects the ditch. Approximately 16,500 m^3 of waste material was excavated from the marsh, and approximately 2,700 m^3 were excavated from the tidal channels. The depth of excavation ranged from approximately 1 to 5 feet in the marsh and from 1 to 4 feet in the tidal channels. The marsh restoration after the excavation consisted of placing backfill over the excavated marsh area, regrading, and revegetating (GeoSyntec Consultants, 1999).

2.3 Available Site Data

This document relies on data that were previously collected for the site. In addition to reviewing documents and data received from NOAA, we also conducted a comprehensive information search to identify any other sources of site-specific information.

Surface water contaminant data, including data on PCBs, mercury, and PAHs, are from Sprenger et al. (1997), Matta et al. (1998), and PTI and CDR (1998). Sediment and soil contaminant data are from two databases provided in CD-ROM format by GeoSyntec Consultants. These databases contain data on contaminants in soil, sediment, surface water, and groundwater collected before, during, and after removal activities at the site. Information on PCB and mercury concentrations in biota (no data are available on PAH concentrations in biota) (summarized in Table 2.2) are from the following sources: Odom (1975), Gardner et al. (1978), Georgia DNR (1996), Sprenger et al. (1997), Kannan et al. (1998), Maruya and Lee (1998a), Matta et al. (1998), and PTI and CDR (1998).

This report is based primarily on information and data that were available as of August 2000. The analyses presented in this report may be updated as new information becomes available.

Table 2.2. Summary of biota samples collected from the vicinity of the LCP site and analyzed for contaminants.

Study	Areas sampled	Organisms sampled[a]	Contaminants
Gardner et al. (1978)	Purvis Creek marsh, Turtle River	**Fish:** black-cheeked tongue fish, common croaker, cutlass fish, gafftopsail catfish, menhaden, sea catfish, silver perch, spot, star drum, trout, weakfish **Invertebrates:** annelid worms, echinoderms, fiddler crab, molluscs, periwinkle snail **Birds:** black-bellied plover, clapper rail, greater yellowlegs, gulls (herring, laughing, and ring-billed), kingfisher, Louisiana heron, pied-billed grebe, red-winged blackbird, snowy egret, spotted sandpiper **Mammals:** cotton rat, Norway rat, opossum, raccoon **Plants:** marsh grass	Mercury
Odom (1975)	Turtle River, Turtle River tributaries	**Invertebrates:** fiddler crab, periwinkle snail, squareback crab **Birds:** clapper rail, king rail, sora rail, Virginia rail	Mercury
Georgia DNR (1996)	Purvis Creek, Purvis Creek marsh, Turtle River	**Fish:** black drum, croaker, flounder, mullet, red drum, sheepshead, spot, spotted seatrout, summer flounder, whiting, yellow tail **Invertebrates:** blue crab, oyster, shrimp	PCBs, mercury
Kannan et al. (1998)	Purvis Creek, Purvis Creek marsh	**Fish:** silver perch, spotted seatrout, striped mullet **Invertebrates:** blue crab **Birds:** boat-tailed grackle, clapper rail, mottled duck, red-winged blackbird **Herpetofauna:** diamondback terrapin	PCBs, organochlorine pesticides
Maruya and Lee (1998a)	Purvis Creek	**Fish:** spotted seatrout, striped fingermullet, striped mullet **Invertebrates:** grass shrimp	PCBs, toxaphene
Matta et al. (1998)	Purvis Creek, Purvis Creek marsh	**Fish:** killifish **Invertebrates:** fiddler crab, oyster	PCBs, mercury, lead
PTI and CDR (1998)	Purvis Creek marsh, Turtle River	**Invertebrates:** fiddler crab, insect, oyster, periwinkle snail, shrimp **Plants:** marsh grass	PCBs, mercury
Sprenger et al. (1997)	Purvis Creek, Purvis Creek marsh, Turtle River	**Fish:** killifish, spot **Invertebrates:** blue crab, brown shrimp, fiddler crab, grasshopper, periwinkle snail **Birds:** clapper rail **Mammals:** cotton rat **Herpetofauna:** diamondback terrapin **Plants:** marsh grass	PCBs, mercury

a. Organism names are as reported by the study authors.

3. Hazardous Substance Pathway Evaluation

This chapter presents a preliminary pathway evaluation for hazardous substances released from the LCP facility. The evaluation focuses on the degree to which available information indicates that PCBs, mercury, and PAHs in the Purvis Creek marsh area originated from the LCP site, and on the potential spatial extent of contamination that resulted from LCP site releases.

3.1 PCB Source Identification

The primary type of PCB used at and released from the LCP site was the commercial mixture Aroclor 1268. Aroclor 1268 is dominated by the more highly chlorinated congeners: octa-, nona-, and deca-chlorobiphenyl congeners constitute 85% of the congeners present (Kannan et al., 1997). Aroclor 1268 is uncommon; it constituted just 0.4% of the commercial PCB sales in the United States from 1957 through 1974 (U.S. EPA, 1976). Therefore, Aroclor 1268 (or a congener pattern similar to Aroclor 1268) in environmental samples collected from the area of LCP site can be traced back to the LCP site with a high degree of confidence (Kannan et al., 1997, 1998).

Several investigations of PCB contamination at and near the Purvis Creek marsh focused on the nature of the PCB congener patterns in sediment and biota and their similarity to Aroclor 1268. Kannan et al. (1997) collected and analyzed sediment samples from the Purvis Creek marsh and from soils excavated from the upland portion of the LCP site. The samples were highly contaminated with PCBs, and octa- and nona-chlorobiphenyls dominated the congener mixture. The authors concluded that the PCBs present in the Purvis Creek marsh originated from the LCP site, because of the similarity of the congener pattern and the strong spatial gradient within the marsh that is consistent with the LCP site as the source. The authors also found some evidence of Aroclor 1260 in site soils and marsh sediments, and concluded that this Aroclor formulation most likely had also been used at the site, although much less of it was released to the environment.

In a follow-up study, Kannan et al. (1998) collected and analyzed fish, birds, blue crabs, and terrapins from the Purvis Creek marsh. The PCB congener pattern observed in the samples was again similar to that of Aroclor 1268, although the relative amount of the most highly chlorinated congeners was somewhat reduced. This reduction was attributed to the decreased membrane permeability of the superhydrophobic congeners. A similar result was found by Maruya and Lee (1998b), who collected and analyzed fish and crustaceans from Purvis Creek.

In addition to the Purvis Creek samples, Maruya and Lee (1998b) also collected several species of fish from Dubignons Creek on the northwest side of Jekyll Island in St. Simons Sound, approximately 25 to 35 km seaward of the LCP site (Figure 3.1). Although total PCB concentrations in these fish were much lower than those in Purvis Creek fish (by an order of magnitude or more), the PCB congeners found in the fish are indicative of Aroclor 1268 and inconsistent with other types of Aroclors. The authors attributed the Aroclor 1268 to the LCP site, and concluded that the fish species sampled (spotted seatrout, red drum, striped mullet, Southern flounder, and Atlantic croaker) had been exposed to the Aroclor 1268 from near the LCP site and had migrated to Jekyll Island. However, they did not have any data from sediment, surface water, or invertebrates from Jekyll Island to confirm that Aroclor 1268 had not migrated via the surface water/sediment pathway. Regardless of the transport pathway, the data from Maruya and Lee (1998b) show that Aroclor 1268 is present in biota 25 to 35 km from the LCP site, indicating that PCBs have been transported at least this distance from the site.

Most of the remainder of the PCB data for the site area are not congener-specific, precluding the identification of the Aroclor 1268 congeners in samples. However, in some investigations the PCBs were identified directly as Aroclor 1268 by comparison to reference standards. For example, in the study by Horne et al. (1999), the PCBs in sediment, fiddler crabs, and marsh periwinkle from Purvis Creek were identified as Aroclor 1268. The State of Georgia laboratory that analyzed fish and shellfish samples from the Turtle River also identified the PCBs in the samples as Aroclor 1268 (Georgia DNR, 1996). Therefore, these data also indicate that the LCP site is the dominant source of PCBs in the samples that have been collected from near Purvis Creek.

3.2 Spatial Patterns of PCB, PAH, and Mercury Contamination in the Turtle River Estuary

Figures 3.2 through 3.4 are maps of the concentrations of total PCBs, mercury, and total PAHs measured in Turtle River area sediments, and Figures 3.5 and 3.6 show total PCB and mercury concentrations measured in Turtle River surface water (PAH data in surface water are insufficient for spatial analysis). All samples shown were collected after the EPA sediment removal. All samples that were reported as not detected are shown on the figures in green, regardless of detection limit (detection limits are discussed in Chapter 4). The data for detected samples shown in the figures were broken into four ranges selected to each incorporate approximately one-quarter of the detected data points. The concentration ranges shown are not intended to imply injury or threshold exceedences, but merely to present the spatial distribution of contaminant concentrations in sediment and water.

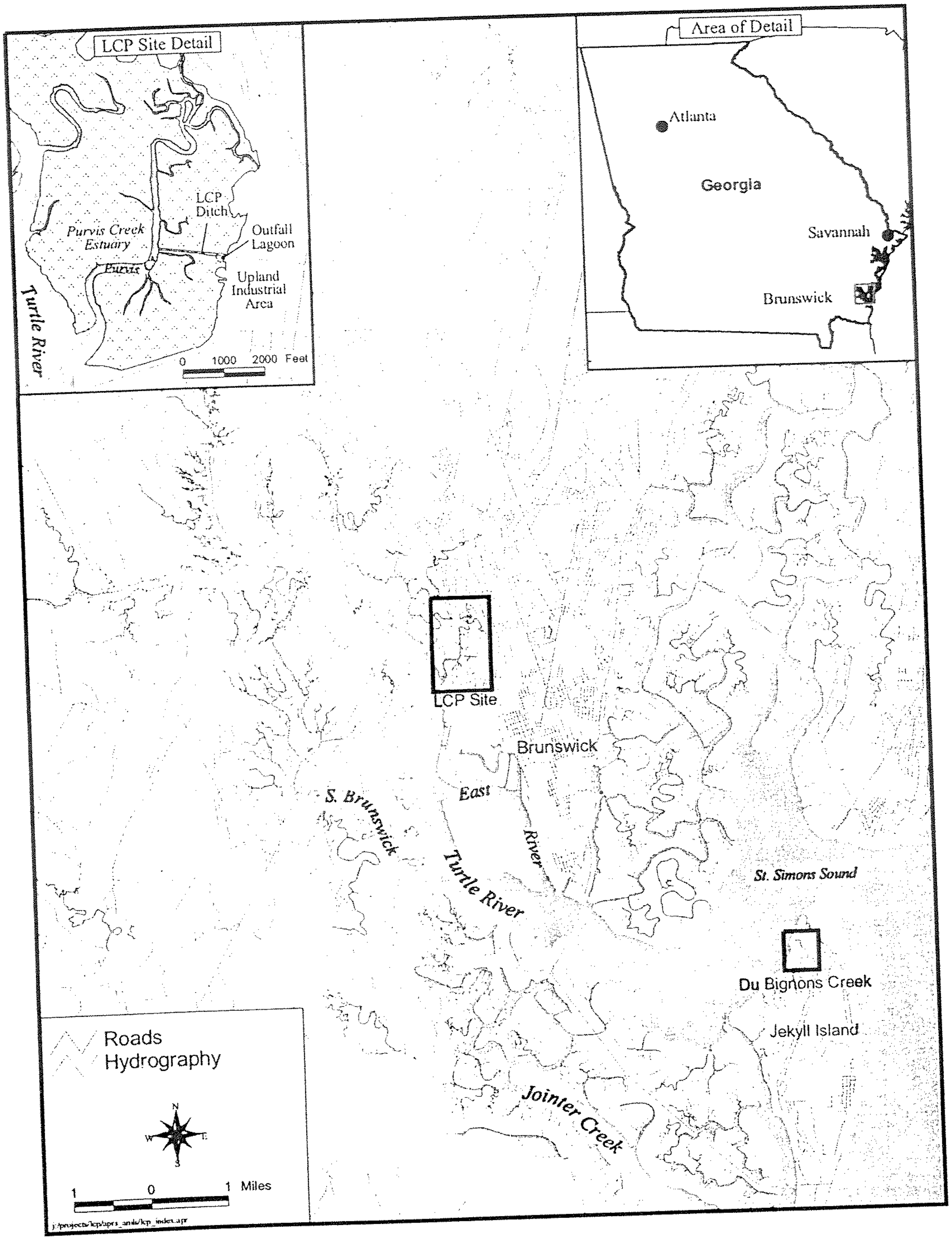

Figure 3.1. Location of Dubignons Creek on Jekyll Island, where PCBs ascribed to the LCP site have been found in fish (Maruya and Lee, 1998b).

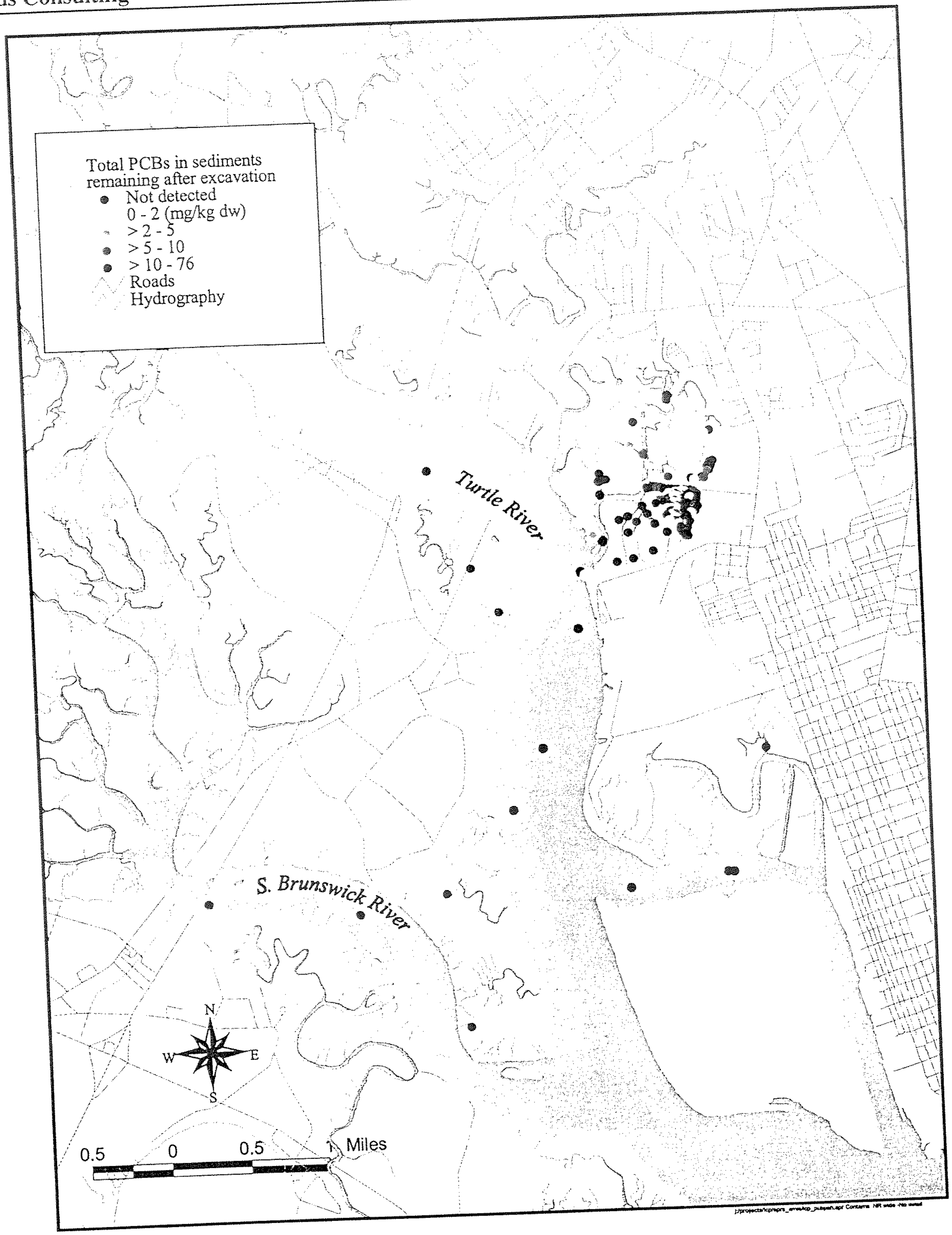

Figure 3.2. Surface sediment total PCB concentrations in the Turtle River estuary. The concentration ranges are for purposes of showing the data distribution and do not represent injury or toxicity thresholds.

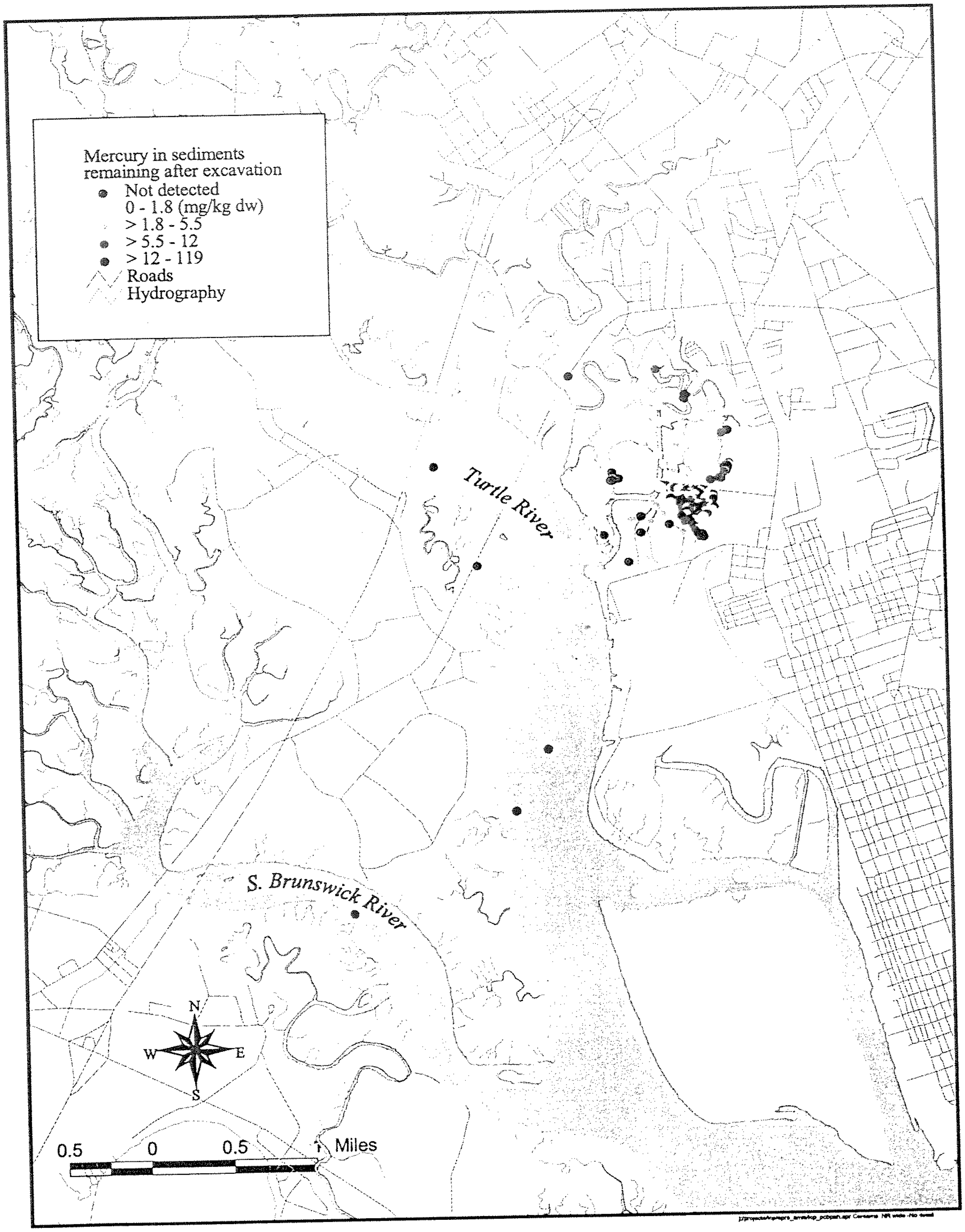

Figure 3.3. Surface sediment total mercury concentrations in the Turtle River estuary. The concentration ranges are for purposes of showing the data distribution and do not represent injury or toxicity thresholds.

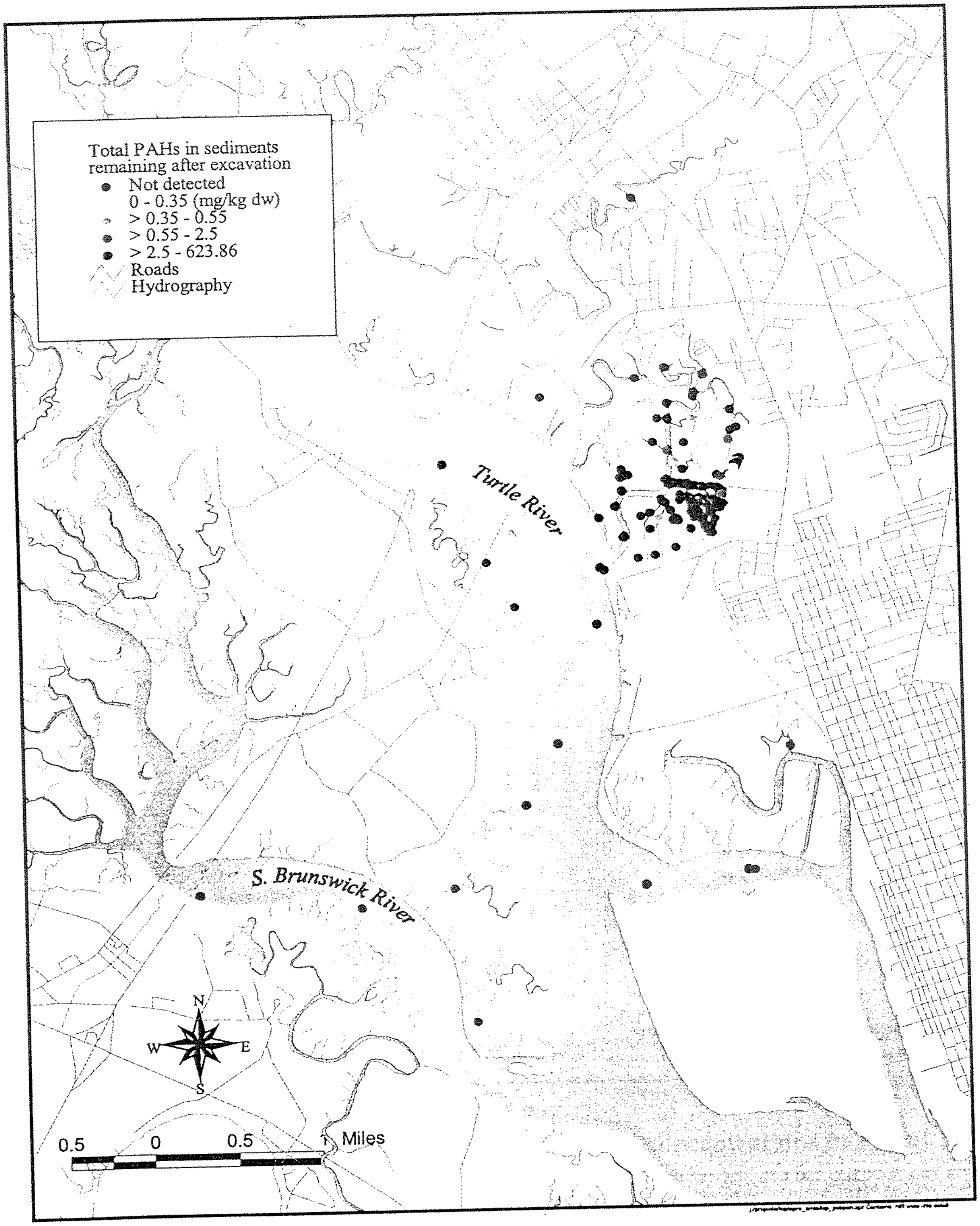

Figure 3.4. Surface sediment PAH concentrations in the Turtle River estuary. The concentration ranges are for purposes of showing the data distribution and do not represent injury or toxicity thresholds.

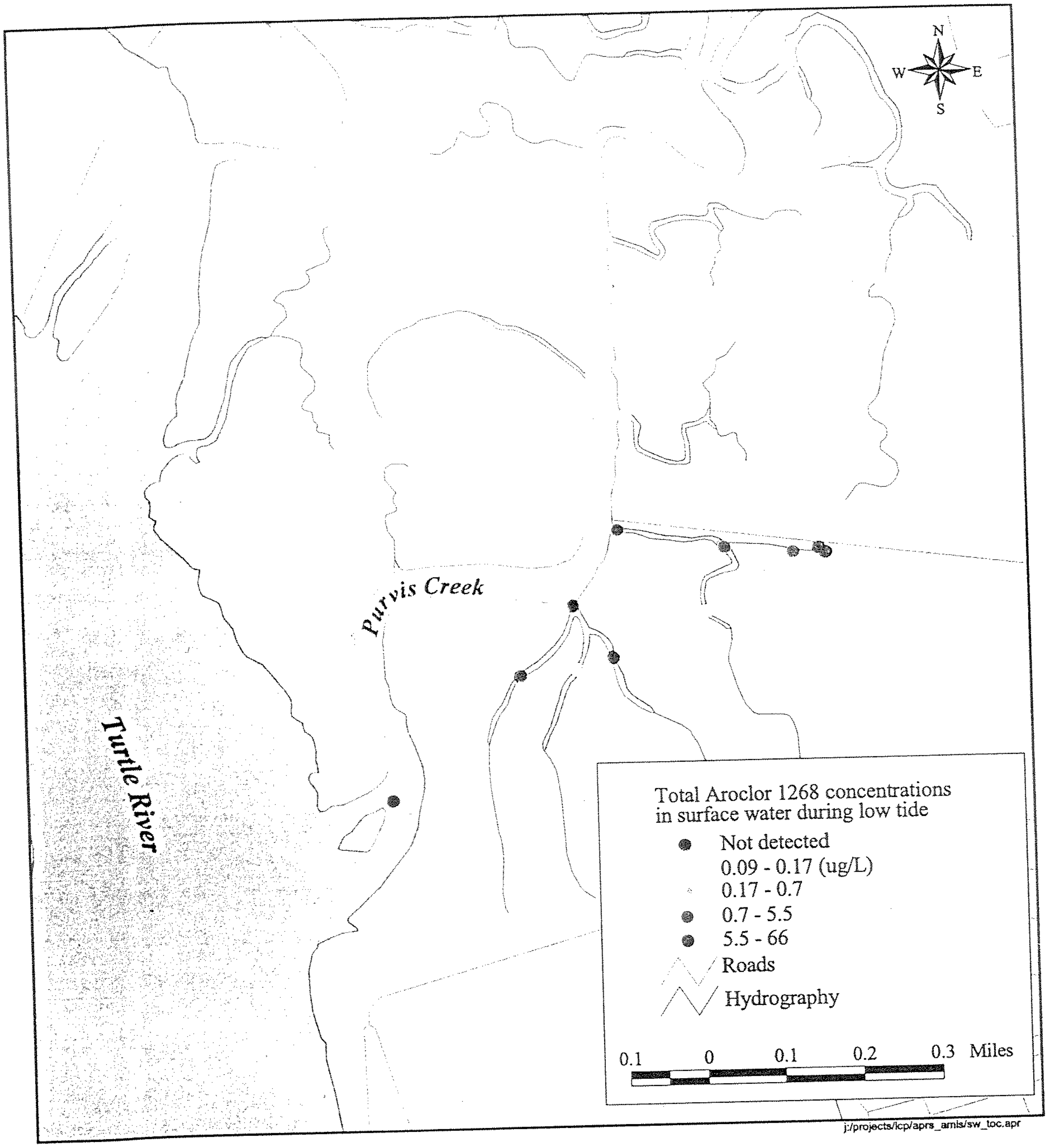

Figure 3.5a. Low tide surface water total Aroclor 1268 concentrations in Purvis Creek, the Purvis Creek marsh, and the Turtle River. The concentration ranges are for purposes of showing the data distribution and do not represent injury or toxicity thresholds.

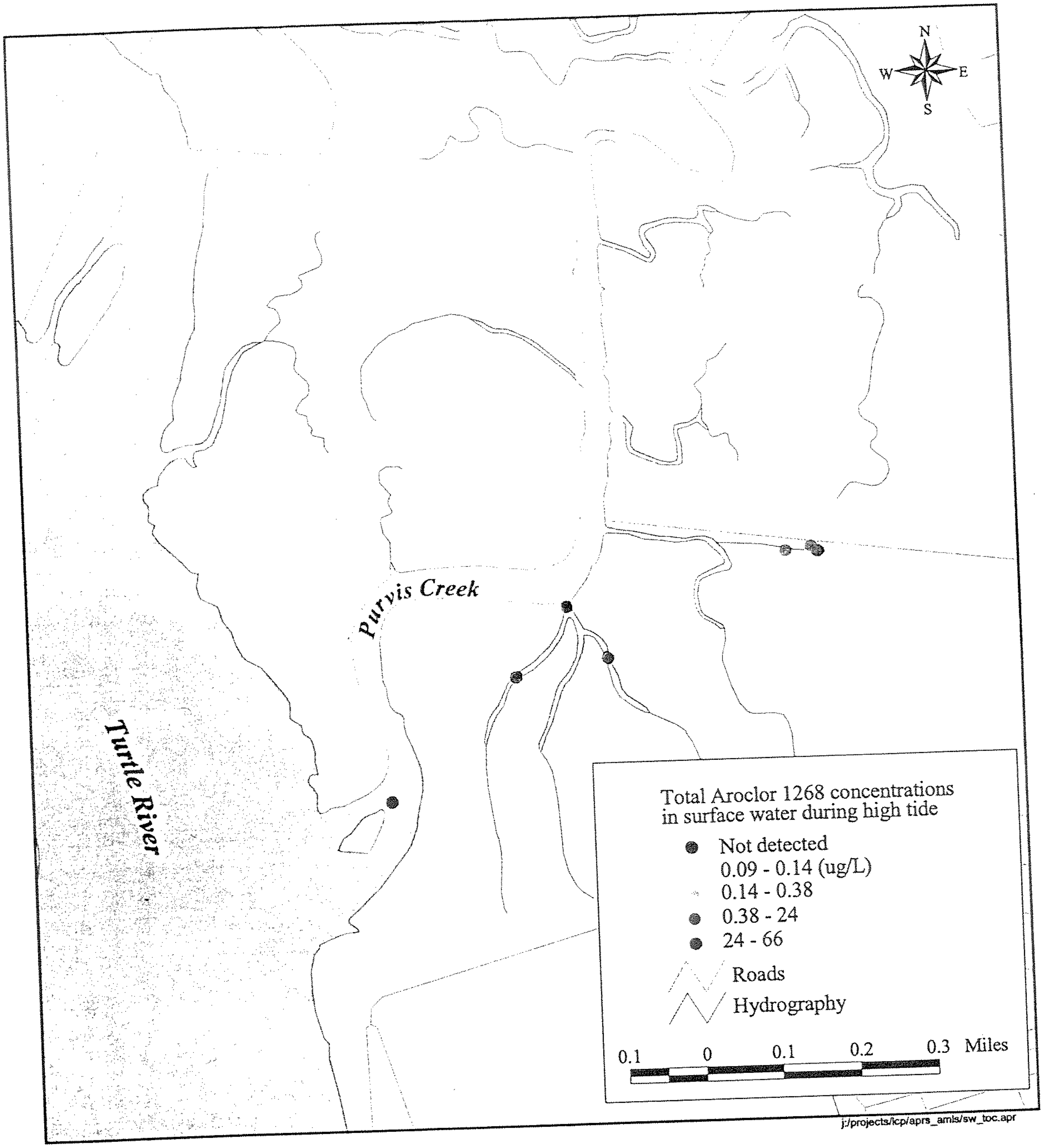

Figure 3.5b. High tide surface water total Aroclor 1268 concentrations in Purvis Creek, the Purvis Creek marsh, and the Turtle River. The concentration ranges are for purposes of showing the data distribution and do not represent injury or toxicity thresholds.

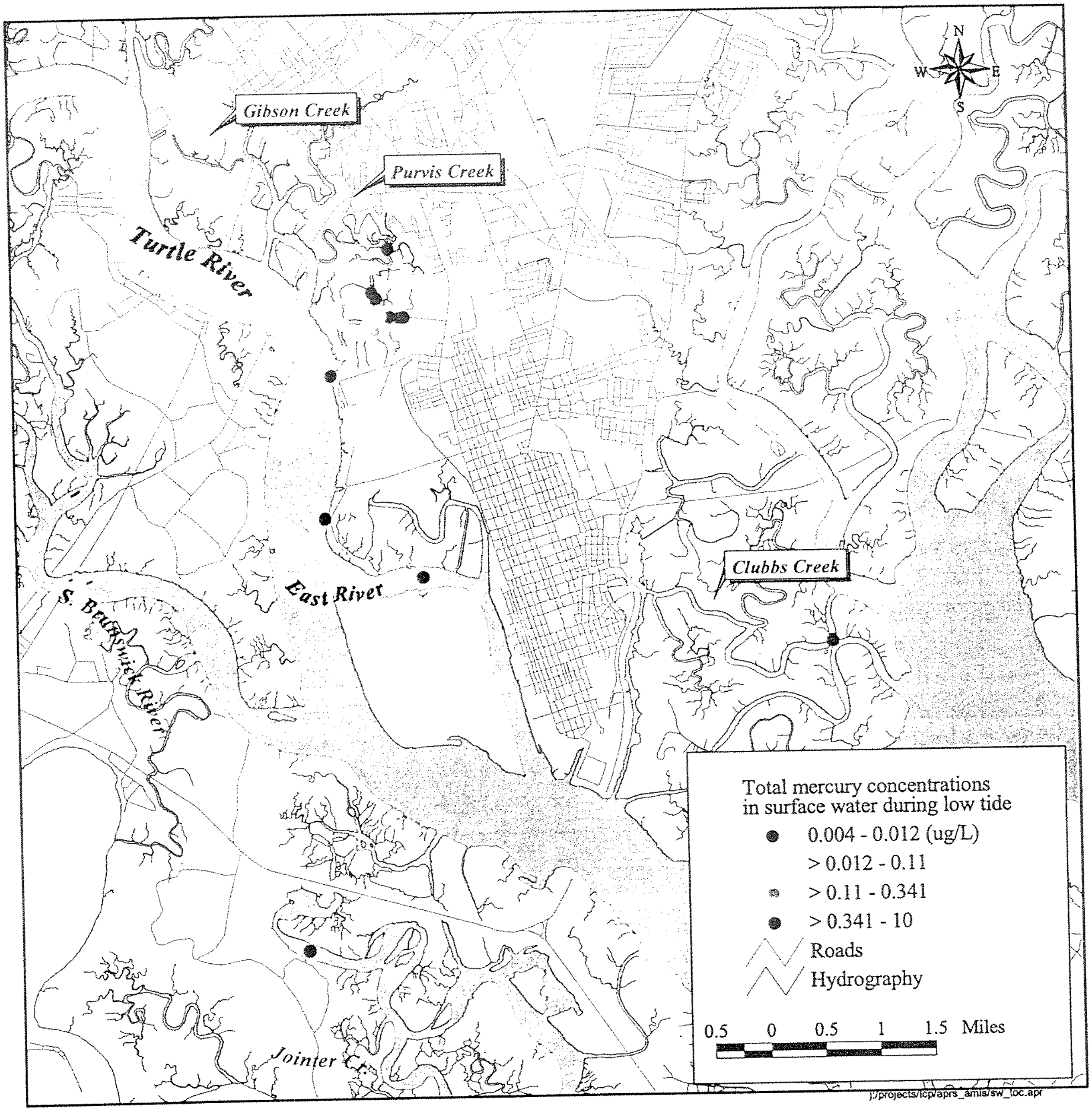

Figure 3.6a. Low tide surface water total mercury concentrations in Purvis Creek, the Purvis Creek marsh, and the Turtle River. The concentration ranges are for purposes of showing the data distribution and do not represent injury or toxicity thresholds.

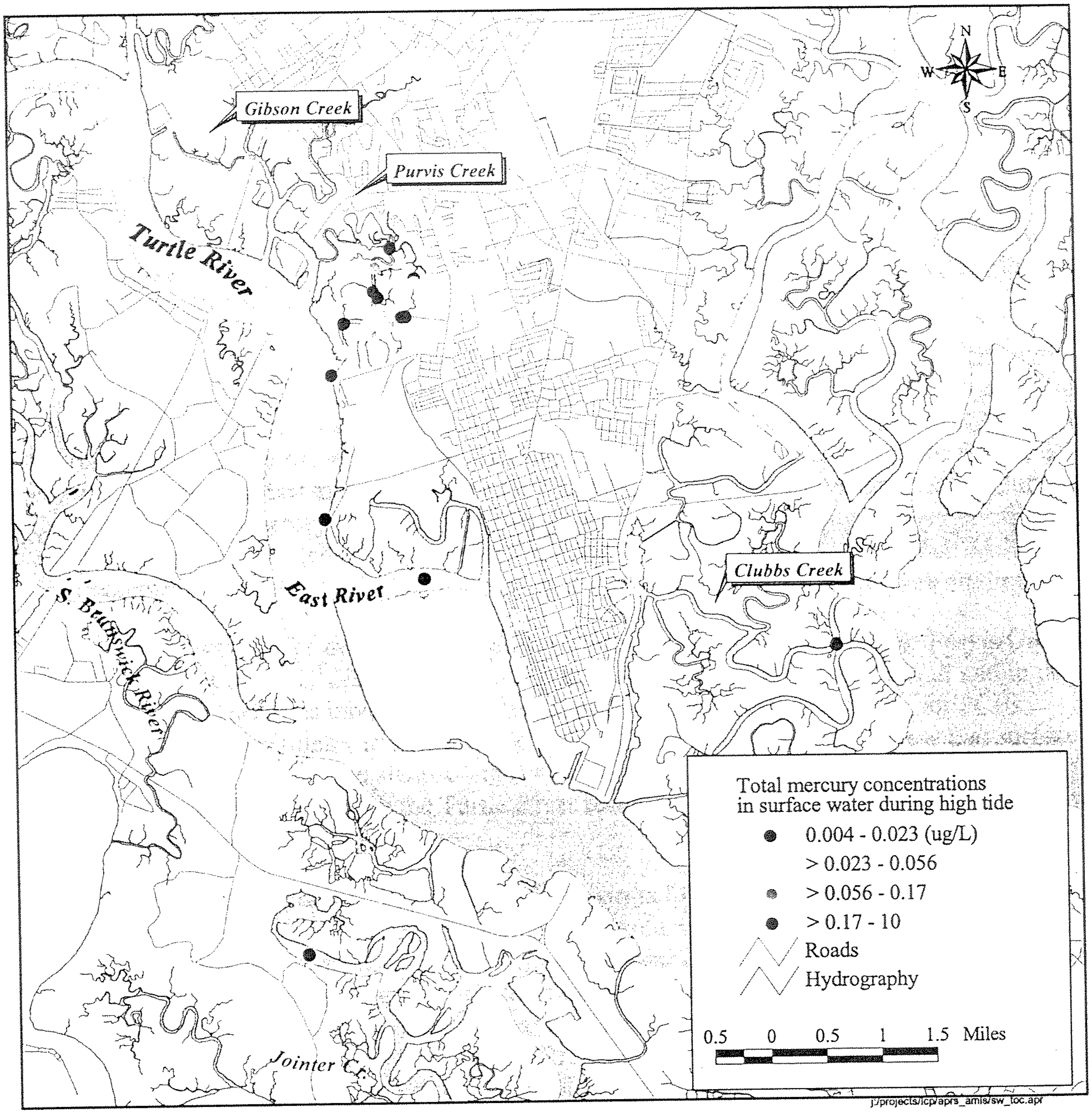

Figure 3.6b. High tide surface water total mercury concentrations in Purvis Creek, the Purvis Creek marsh, and the Turtle River. The concentration ranges are for purposes of showing the data distribution and do not represent injury or toxicity thresholds.

Figures 3.2 through 3.4 show that most area sediment samples have been collected from the Purvis Creek marsh, and few samples have been collected from the Turtle River or other tributaries to the Turtle River (including the South Brunswick River). The data from the Purvis Creek marsh show a strong spatial concentration gradient that declines with distance from the LCP site, which is consistent with the LCP site as the dominant source to the marsh. The concentrations of PCBs, PAHs, and mercury in most of the sediment samples from areas other than the Purvis Creek marsh are below the analytical detection limits used. Therefore, this information primarily emphasizes the high contamination in the Purvis Creek marsh and its consistency with the LCP site as the source, and is of only limited value in delineating the spatial extent of contamination beyond the marsh.

Nevertheless, the three Turtle River or South Brunswick River sediment samples with detectable PCBs were all collected right at the mouth of Purvis Creek (Figure 3.2), as were two of the three Turtle River or South Brunswick River samples with detectable PAHs (Figure 3.4). Therefore the data provide some evidence that PCBs and PAHs are entering the Turtle River from the Purvis Creek marsh. For mercury (Figure 3.3), highest concentrations are again in the Purvis Creek marsh, and mercury was detected in approximately 70% of the Turtle River or other tributary samples collected. No clear spatial relationship of mercury contamination with distance or direction from the Purvis Creek marsh is evident in the data, although the data are limited.

Figure 3.5 shows that the available surface water data on PCBs are restricted to Purvis Creek and the Purvis Creek marsh. PCB concentrations at the site are highest near the outfall and in the LCP ditch. More samples have been analyzed for mercury concentrations than for PCBs, including several locations within the Turtle River (Figure 3.6). These data show that surface water mercury concentrations are also highest at the site near the outfall and in the LCP ditch, and the few samples taken from the Turtle River reveal much lower mercury concentrations with no discernible spatial patterns.

Mean mercury concentrations measured by the Georgia DNR (1996) in blue crabs of the Turtle River estuary area are shown in Figure 3.7. Mean mercury concentrations are highest in crabs from the Turtle River near Purvis Creek (3.12 mg/kg ww), followed by concentrations in crabs from Purvis Creek (1.45 mg/kg ww) and from Gibson Creek (1.69 mg/kg ww). Mercury concentrations in blue crabs from other areas of the Turtle River estuary are all much lower, and are fairly consistent with each other (all from 0.11 to 0.32 mg/kg ww). These data indicate that blue crabs from near the site are highly contaminated with mercury, and that the contamination extends into the Turtle River and the nearby Gibson Creek area (possible via crab migration).

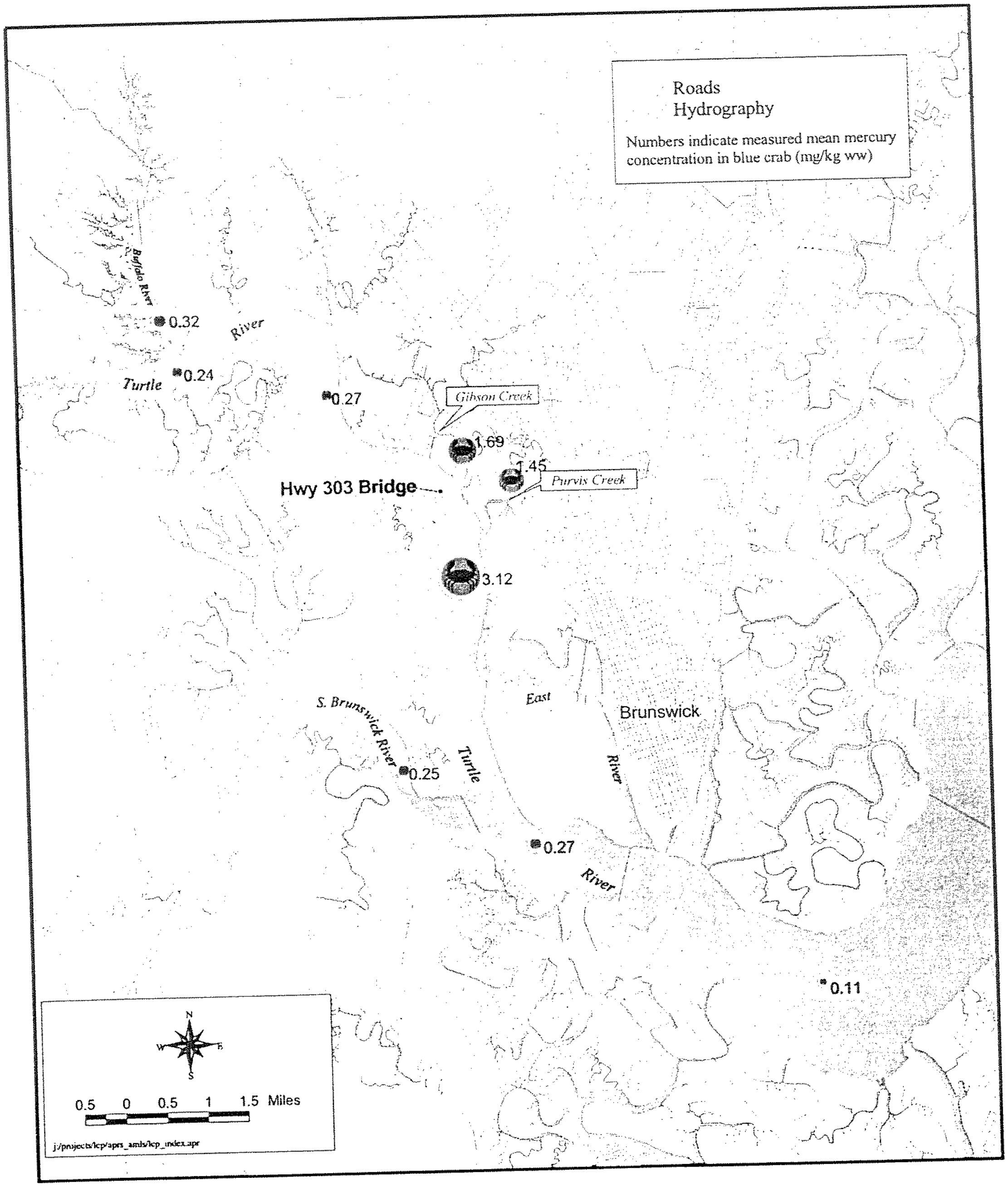

Figure 3.7. Mean mercury concentrations in blue crabs collected by the Georgia DNR (1996). The area of each symbol is proportional to the mean mercury concentration at that sampling location.

3.3 Conclusions

All the available data on PCB congener patterns and Aroclor mixtures indicate that the LCP site is the dominant source of PCBs in sediment and biota in Purvis Creek and the Purvis Creek marsh. The PCB congener patterns in fish from the Turtle River and from a creek on Jekyll Island, approximately 25 to 35 km from the LCP site, are also consistent with the unique type of PCBs released from the LCP site.

The extensive data on the spatial patterns of PCB, mercury, and PAH contamination in Purvis Creek and the Purvis Creek marsh are consistent with the LCP site being the dominant source of these contaminants to these areas. However, the available data relevant to defining the spatial extent of contamination in the Turtle River are limited. The data that are available indicate that PCB, mercury, and PAH concentrations are highest in the Turtle River near the mouth of Purvis Creek, consistent with the LCP site being the dominant source of these contaminants to at least this area of the river. Available data on mercury in blue crabs indicate that mercury contamination from the site is highest in Purvis Creek, Gibson Creek, and the Turtle River near these creeks.

4. Surface Water and Sediment Standards/ Criteria Exceedences

This chapter presents a comparison of PCB and mercury concentrations measured in surface water and sediment with relevant standards or criteria that are identified in the DOI NRDA regulations at 43 CFR Part 11. The surface water resources evaluated in this preliminary assessment include the surface waters and sediments (including bed, bank, and shoreline sediments) of Purvis Creek, the Purvis Creek marsh, and the Turtle River.

4.1 Relevant Standards and Criteria

The DOI regulations include the following definition of injury to surface water resources:

> Concentrations and duration of hazardous substances in excess of applicable water quality criteria established by Section 304(a)(1) of the Clean Water Act (CWA), or by other federal or state laws or regulations that establish such criteria, in surface water that before the . . . release met the criteria and is a committed use as habitat for aquatic life, water supply, or recreation. The most stringent criterion shall apply when surface water is used for more than one of these purposes [43 CFR § 11.62(b)(1)(iii)].

Table 4.1 lists criteria and standards applicable to an evaluation of injury to surface waters at the LCP site. Pursuant to Section 304 of the Clean Water Act (CWA), the U.S. EPA has established ambient water quality criteria (AWQC) for the protection of aquatic life. The Georgia Rules and Regulations for Water Quality Control set general standards for all waters of the state and include separate listings for coastal and marine waters. The AWQC listed in Table 4.1 are those that were in effect at the time that the surface water data for the LCP site were collected (1995 through 1997, as described in Section 4.2.1). Although the AWQC for both PCBs and mercury were changed in 1999 to reflect an updated analysis of the underlying toxicity data by U.S. EPA, the criteria that were in place at the time of sample collection are used to evaluate surface water injuries pursuant to the injury definition listed above. The updated AWQC are used in Chapter 5 to evaluate the potential for measured surface water concentrations to cause toxicity to aquatic biota. As Table 4.1 shows, the AWQC in place in 1995-1997 and the Georgia standards are the same for both PCBs (0.014 µg/L total PCBs) and mercury (0.025 µg/L total mercury).

The Georgia Rules and Regulations identify seven different Aroclors to which the PCB standard applies (the standard applies to the sum of the Aroclors). However, Aroclor 1268, which is the

Table 4.1. State and federal water quality standards and criteria for PCBs and mercury in surface water.[a]

Source	PCB standard or criterion (µg/L)	Mercury standard or criterion (µg/L)
Georgia Rules and Regulations for Water Quality Control (for coastal and marine estuarine waters)[b]	0.014[c]	0.025
U.S. EPA national recommended water quality criteria (AWQC), chronic exposure value (CCC) (for saltwater)	0.014[c]	0.025

a. Criteria are expressed as total concentration.
b. Georgia DNR, 2000, Chapter 391-3-6-.03-5.
c. Applies to seven Aroclors but not specifically to Aroclor 1268.

PCB mixture released at the LCP site, is not specifically identified, most likely because of its uncommon use. Nevertheless, for the purposes of this preliminary injury evaluation, we assume that the regulations for other Aroclors may be applied to Aroclor 1268, since its lack of inclusion in the regulations is most likely a result of its rarity in the environment rather than a specific regulatory exclusion.

NRDA injury definitions apply to waters with a committed use as habitat for aquatic life, water supply, or recreation. The waters of Purvis Creek, the Turtle River, and the Brunswick River have a specific water use classification of "fishing: propagation of fish, shellfish, game, and other aquatic life" and therefore have a committed use as habitat for aquatic life (Georgia DNR, 2000, Chapter 391-3-6-.03-13). The Georgia DNR fishing water use classification also includes secondary contact recreation in and on the water. The surface waters in St. Simons Sound are additionally classified as waters generally supporting shellfish, where "shellfish" refers to clams, oysters, scallops, mussels, and other bivalve mollusks. Waters with this shellfish classification include Turtle River from the confluence with Buffalo River to South Brunswick River, including Purvis Creek (but excluding Gibson Creek), and Brunswick River from the confluence with Turtle River and South Brunswick River to St. Simons Sound.

4.2 Surface Water Standards and Criteria Exceedences

4.2.1 Data sources

The following data on concentrations of PCBs and mercury in site area surface waters are available:

1. Data collected in 1995 for the ERA prepared by the U.S. EPA (Sprenger et al., 1997). Samples were collected from three sites in Purvis Creek and four sites in the Purvis Creek marsh and analyzed for PCBs (total only) and mercury (total and dissolved).

2. Data collected in 1996 for the ERA prepared by PTI and CDR (PTI and CDR, 1998). Samples were collected from four sites in Purvis Creek, one site in the marsh, and two sites in the Turtle River. They were analyzed for PCBs (dissolved) and mercury (total and dissolved).

3. Data collected in 1997 for the LCP Chemical Site Monitoring Study prepared by NOAA, U.S. EPA, and EVS Environment Consultants (Matta et al., 1998). Samples were collected from one site in Purvis Creek, five sites in the marsh, and one site in the Turtle River and analyzed for PCBs (total only) and mercury (total and dissolved).

4. Data collected in 1998-1999 during the EPA removal activities (GeoSyntec Consultants, 1999). Samples were collected from the LCP ditch and analyzed for total PCBs and mercury.

4.2.2 PCB concentrations in surface water

Surface water total PCB concentrations measured in Purvis Creek and in the marsh are compared with the Georgia water quality standard and the AWQC in Table 4.2.[1] Surface water PCB concentration data are available for nine samples taken from Purvis Creek and 14 samples from

Table 4.2. Exceedences of Georgia PCB surface water standard and AWQC.

Location	No. of samples	Nondetects (detection limit)	Range of detected values (µg/L)	Detected values exceed Georgia standard and AWQC of 0.03 µg/L?
Purvis Creek	9	5 (0.2 µg/L)	0.17-5.5	Yes for all
Purvis Creek marsh	14	7 (0.2 µg/L)	0.2-66	Yes for all
Turtle River	2	2 (0.2 µg/L)	NA	NA

All concentrations shown are for total PCBs (see footnote 1 in text).

1. The dissolved PCB samples collected by PTI and CDR (1998) are not shown in the table because the very high detection limit of 200 µg/L in the study makes the data essentially unusable.

the marsh, all collected in 1995 and 1997. PCBs were detected in four of the samples from the creek, at concentrations up to 5.5 μg/L (total), and in seven of the samples from the marsh, at concentrations up to 66 μg/L (total). All of the detected concentrations greatly exceeded the Georgia standard and the AWQC. PCBs were not detected in five samples from the creek and seven samples from the marsh, but the high detection limit (0.2 μg/L) precludes any determination as to whether the standard or AWQC was exceeded.

Two samples from the Turtle River were analyzed for total PCBs. PCBs were not detected in either of the samples, at a detection limit of 0.2 μg/L. Again, the high detection limit means that no conclusion can be drawn as to whether PCBs exceeded the Georgia standard or AWQC in the samples.

4.2.3 Mercury concentrations in surface water

Surface water mercury concentrations in Purvis Creek, Purvis Creek marsh, and the Turtle River are compared to the Georgia standard and the AWQC in Table 4.3. Total mercury was detected in all 13 of the Purvis Creek samples, at concentrations up to 0.5 μg/L. Total mercury exceeded the Georgia standard and AWQC of 0.025 μg/L in 12 of the 13 samples.

In the Purvis Creek marsh, total mercury was detected in all 15 samples collected, at a maximum concentration of 10 μg/L. Total mercury concentrations in all 15 samples exceeded the Georgia standard and AWQC of 0.025 μg/L. The two samples from the Turtle River had 0.0094 and 0.01 μg/L dissolved mercury, which are both below the Georgia standard and AWQC. However, the Turtle River samples were analyzed for dissolved mercury only, and total mercury concentrations are most likely higher than the dissolved concentrations.

Table 4.3. Exceedences of mercury surface water standard and AWQC.

Location	Analysis type	No. of samples	Nondetects (detection limit)	Range of detected values (μg/L)	Detected values exceed Georgia standard and AWQC of 0.025 μg/L?
Purvis Creek	Total	13	0	0.022-0.500	Yes for 12 samples
Purvis Creek marsh	Total	15	0	0.045-10	Yes for all 15 samples
Turtle River	Dissolved[a]	2	0	0.0094-0.01	No for both samples

a. Standard and AWQC apply to total mercury, but only dissolved results were obtained.

4.2.4 Contaminant concentrations during removal activities

During the removal actions between January 1998 and July 1999, water samples were collected from the LCP ditch and analyzed for total concentrations of mercury and PCBs. PCB concentrations ranged from undetected (at a detection limit of 1 μg/L) up to 19.8 μg/L in a sample collected on July 27, 1998. Mercury concentrations ranged from undetected (at a detection limit of 1 μg/L) up to 29.9 μg/L in a sample collected on March 18, 1998. Both the detection limits and the maximum concentrations measured are well above the applicable Georgia standard and AWQC.

However, these data may not be representative of contaminant concentrations normally present in the ditch, since removal activities may have temporarily increased contaminant concentrations in ditch water. Nevertheless, these data document the high concentrations of PCBs and mercury present at the site, and are additional evidence that standards and criteria have been exceeded in waters near the site.

4.3 Conclusions

Relevant state standards and federal criteria have been greatly exceeded in the waters of Purvis Creek and the Purvis Creek marsh. Measured concentrations of PCBs and mercury have exceeded Georgia state standards and U.S. EPA AWQC by many orders of magnitude. Exceedences have not been documented in the Turtle River, but only limited surface water sampling has been conducted, using PCB detection limits that are much higher than the Georgia standards and AWQC, and analyzing only dissolved mercury.

These data show that surface water resource has been injured by exceedences of relevant state standards and federal criteria for the protection of surface water.

Table 4.3. Exceedences of mercury surface water standard and AWQC.

Location	Analysis type	No. of samples	Nondetects (detection limit)	Range of detected values (μg/L)	Detected values exceed Georgia standard and AWQC of 0.025 μg/L?
Purvis Creek	Total	13	0	0.022-0.500	Yes for 12 samples
Purvis Creek marsh	Total	15	0	0.045-10	Yes for all 15 samples
Turtle River	Dissolved[a]	2	0	0.0094-0.01	No for both samples

a. Standard and AWQC apply to total mercury, but only dissolved results were obtained.

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However, these data may not be representative of contaminant concentrations normally present in the ditch, since removal activities may have temporarily increased contaminant concentrations in ditch water. Nevertheless, these data document the high concentrations of PCBs and mercury present at the site, and are additional evidence that standards and criteria have been exceeded in waters near the site.

4.3 Comparison of Sediment PCB Concentrations with TSCA Regulations

Figures 4.1 and 4.2 show the distribution of sediment samples that exceed the TSCA 50 mg/kg threshold for PCBs in marsh sediments. Figure 4.1 shows exceedences in sediments that were later excavated during removal activities, and Figure 4.2 shows exceedences in sediments that remain in the marsh. As discussed in Chapter 2, excavation took place in the LCP ditch, in the tidal channel intersecting the ditch, and in the marsh area east of the tidal channel. Although PCB concentrations in sediments that were later excavated frequently exceeded the threshold, only one sample of those collected in areas that remain after excavation had a PCB concentration exceeding 50 mg/kg.

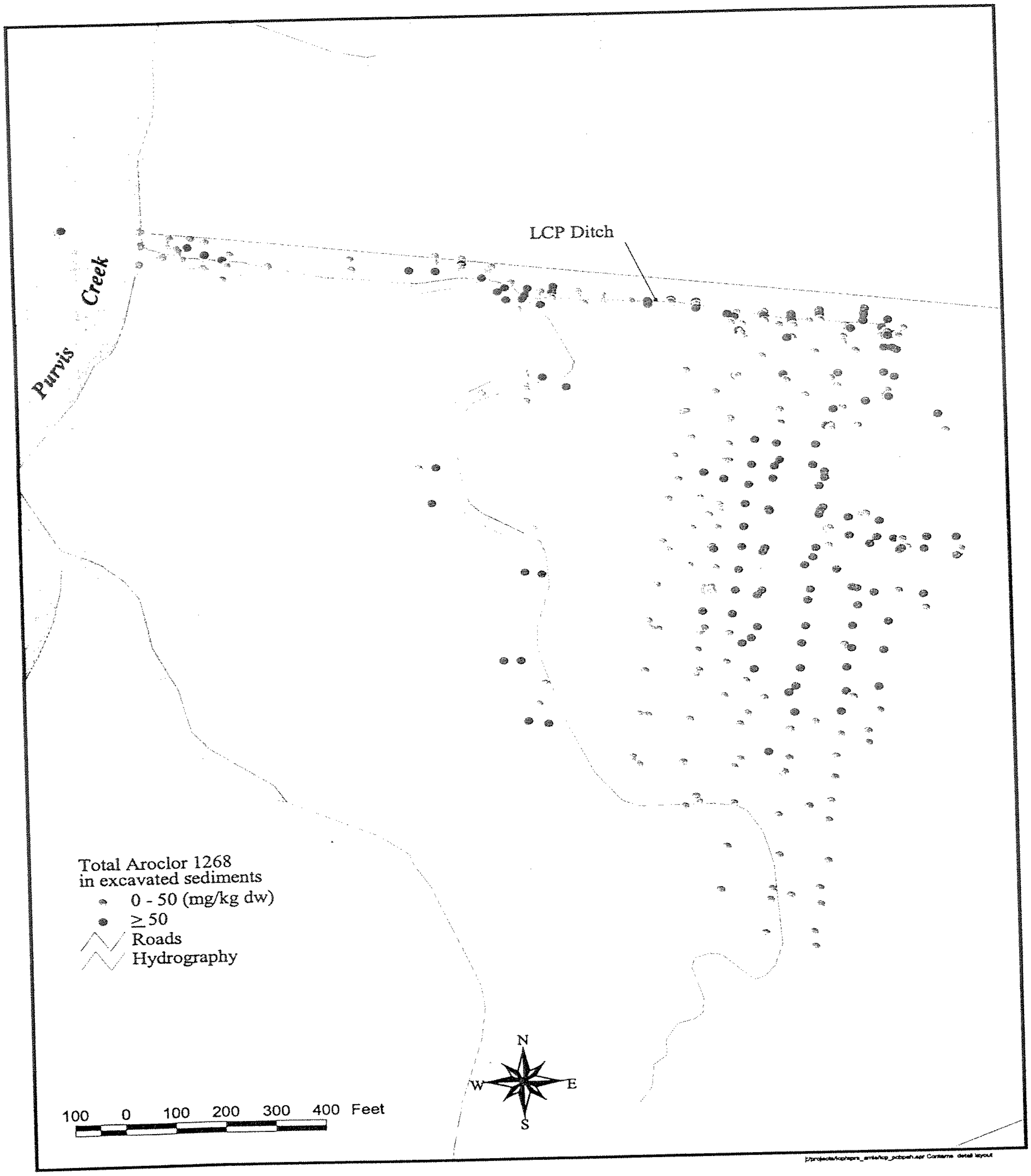

Figure 4.1. Exceedences of the TSCA 50 mg/kg threshold for PCBs in marsh sediments that were later excavated during removal.

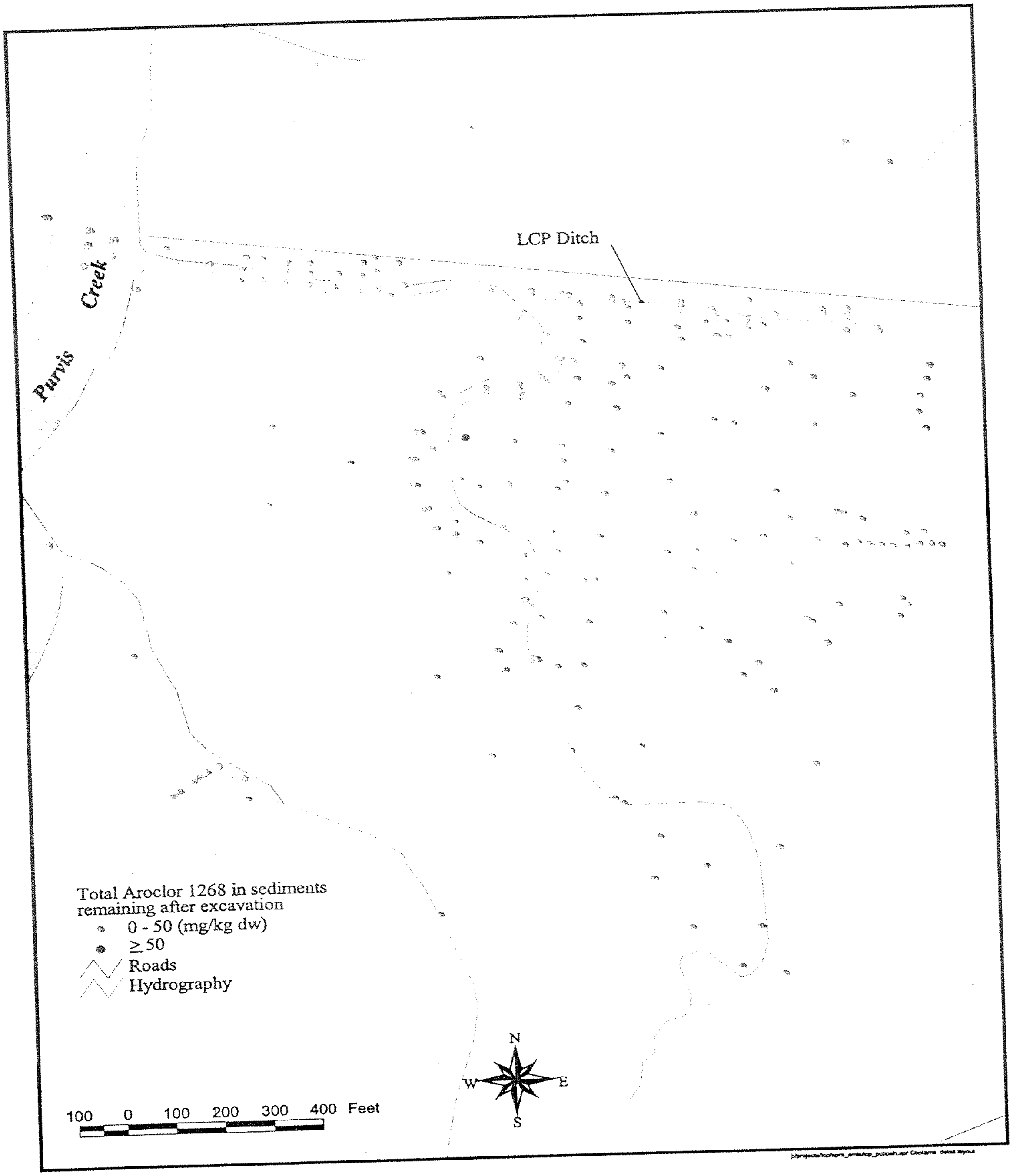

Figure 4.2. Exceedences of the TSCA 50 mg/kg threshold for PCBs in marsh sediments that remain in the marsh. These samples were taken either in unexcavated areas or after excavation in excavated areas.

4.4 Conclusions

Relevant state standards and federal criteria have been greatly exceeded in the waters of Purvis Creek and the Purvis Creek marsh. Measured concentrations of PCBs and mercury have exceeded Georgia state standards and U.S. EPA AWQC by many orders of magnitude. Exceedences have not been documented in the Turtle River, but only limited surface water sampling has been conducted, using PCB detection limits that are much higher than the Georgia standards and AWQC, and analyzing only dissolved mercury. PCB concentrations in many of the samples collected from areas of the Purvis Creek marsh that were excavated by U.S. EPA exceeded the 50 mg/kg TSCA threshold for hazardous waste disposal, but this threshold is exceeded in only one sample from areas that remained after excavation.

These data show that surface water resource has been injured by exceedences of relevant state standards and federal criteria for the protection of surface water.

5. Adverse Effects Injuries to Biological Resources

This chapter presents a preliminary evaluation of the potential for PCBs, mercury, and PAHs released from the LCP to cause adverse toxic effects to exposed biological resources.

The exposure of biota in Purvis Creek, the Purvis Creek marsh, and the Turtle River to elevated concentrations of LCP site contaminants has been well documented (Section 2.3). Elevated concentrations of PCBs and mercury have been measured in fish, birds, terrapins, crabs, other invertebrates, and plants, thus confirming that these biota are both exposed to and bioaccumulating LCP site contaminants in surface water and sediment. As described in Chapter 3, the PCBs in fish from as far as 25-35 km from the LCP site have been identified as the same rare PCB mixture that was released from the LCP site. Studies of PCB contamination in biota in the area have concluded that ingestion of contaminated prey items is the primary exposure route for predatory fish and birds (Kannan et al., 1998; Maruya and Lee, 1998b).

This chapter evaluates the likelihood that the exposure of biota to elevated concentrations of site contaminants has caused and continues to cause adverse effects. First, the ERAs that were conducted at the site are reviewed in Section 5.1. Section 5.2 describes studies that investigated the direct toxicity of Purvis Creek or marsh surface water and/or sediment. Finally, in Section 5.3 the concentrations of contaminants that have been measured in surface water, sediment, and biota are compared to concentrations that have been shown to cause toxicity in laboratory studies or at other sites.

5.1 Ecological Risk Assessments Conducted at LCP Site

Two separate ERAs have been conducted at the LCP site. The first (Sprenger et al., 1997) was conducted by the Environmental Response Team (ERT) of the U.S. EPA, and the second (PTI and CDR, 1998) was conducted for Allied Signal, Inc. The assessment endpoints used in both ERAs and the conclusions of each are compared in Table 5.1.

Sprenger et al. (1997) concluded that threats to more assessment endpoints exist than did PTI and CDR (1998). Sprenger et al. (1997) found that maintenance of ecological health of the salt marsh was threatened by site contamination. The conclusion was based on sediment concentrations of mercury, PCBs, and zinc that exceed ecological benchmarks and on the results of site sediment toxicity tests. In addition, a benthic invertebrate community study conducted as part of the ERA found that the community shifts from an even distribution of surface and subsurface feeders in

Table 5.1. Assessment endpoints and conclusions of Sprenger et al. (1997) and PTI and CDR (1998) ERAs.

Assessment endpoint	Sprenger et al. (1997) conclusion	PTI and CDR (1998) conclusion
Maintenance of ecological health and function of the salt marsh community	Risk based on concentrations in sediment and risk to benthic organisms	No risk
Protection of the long-term health and reproductive capacity of aquatic reptiles	No risk	No risk
Protection of long-term health and reproductive capacity of omnivorous mammals	Risk based on food chain modeling for raccoon	No risk
Protection of long-term health and reproductive capacity of piscivorous mammals	Risk based on food chain modeling for river otter	Risk based on food-web modeling for river otter
Protection of long-term health and reproductive capacity of avian species	Risk based on food chain modeling for clapper rail and marsh wren	No risk
Protection of health and reproductive capacity of fishery resources	Risk based on killifish body burdens and sediment toxicity tests with medaka embryos	No risk
Protection of the fish nursery function	Risk based on sediment toxicity tests with medaka embryos	No risk
Protection of individual ridley turtle	No risk	No risk
Protection of individual green turtle	No risk	No risk
Protection of individual wood stork	No risk	No risk
Protection of individual manatee	No risk	No risk
Protection of individual shortnose sturgeon	Risk based on killifish body burdens	No risk

Sources: Sprenger et al., 1997; PTI and CDR, 1998.

cleaner areas to a community dominated by surface feeders in the more contaminated areas (Horne et al., 1999). Sprenger et al. (1997) also concluded that a risk to benthic organisms existed based on reduced lipid content and elevated PCB body burdens exceeding toxic thresholds in fiddler crabs collected from the estuary. Sprenger et al. (1997) concluded that an acute exposure risk from site contamination threatens omnivorous and piscivorous mammals, avian species, fishery resources and fishery nursery functioning, and the endangered shortnose sturgeon. No potential risk was found to threatened or endangered sea turtles, wood storks, or manatees.

In contrast, PTI and CDR (1998) conclude that there are no discernible site-related effects to maintenance of ecological health of the salt marsh; to long-term health and reproductive capacity of aquatic reptiles, omnivorous mammals, finfish, or birds; or to protection of fishery nursery function, threatened or endangered marine turtles, wood storks, West Indian manatees, or shortnose sturgeon. PTI and CDR (1998) conclude that the only potentially unacceptable risk associated with contaminant exposure is a chronic PCB exposure risk to piscivorous mammals, based on food-web modeling using the river otter as the endpoint.

At least some of the differences in conclusions between the two ERAs may stem from differences in some of their methods. For example, PTI and CDR (1998) attempted to develop toxicity reference values specific to Aroclor 1268, whereas Sprenger et al. (1997) relied on values available in the literature for Aroclor 1254. Another difference is that PTI and CDR (1998) apply a fractional area use factor to account for organisms foraging in areas besides the site, whereas Sprenger et al. (1997) assumed an area use factor of one. It should be noted that methods and conclusions of the ERA conducted by PTI and CDR (1998) have been disputed by the regulatory agencies.

5.2 Toxicity and Adverse Effect Studies

This section describes the field and laboratory studies that have been conducted at the site to investigate whether site contamination is causing adverse effects to biota. The studies are grouped into three categories: toxicity studies in which organisms are exposed to site surface water, porewater, or sediment in a controlled laboratory setting (Section 5.2.1); field studies on contaminant bioaccumulation and toxicity using oysters (Section 5.2.2); and benthic macroinvertebrate community studies (Section 5.2.3). All of the studies reported here were conducted prior to completion of EPA's emergency removal action in the marsh.

5.2.1 Surface water and sediment toxicity studies

Surface water tests using mysid shrimp and sheepshead minnows

PTI and CDR (1998) conducted toxicity tests on site surface water using mysids and sheepshead minnows. The endpoints evaluated were mysid survival, growth, and fecundity, and sheepshead minnow survival and teratogenicity. Juvenile mysids or embryonic/larval sheepshead minnows were exposed for nine days to ambient surface water collected from nine sites in tidal channels of the estuary and two reference sites. Laboratory controls were also used. Exposure durations were seven days for mysids and nine days for minnows.

The results of the toxicity tests are summarized in Table 5.2. No statistically significant differences in results between site locations and reference locations were observed for any of the endpoints. All female mysids that survived exposure had eggs in their oviducts or brood sacs. A single deformed sheepshead minnow (out of 660 used in the experiment) was noted (PTI and CDR, 1998).

Table 5.2. Results of water column exposure experiments with mysids and sheepshead minnows by PTI and CDR (1998).

Sampling station	7-day mysid survival (%)	7-day mysid growth[a] (mg/kg dw)	9-day sheepshead minnow survival (%)
LCP Ditch	82	0.220	67
Middle Purvis Creek	85	0.262	67
Upper Purvis Creek	85	0.230	67
Purvis Creek headwaters	88	0.234	77
Lower Purvis Creek	88	0.259	72
Turtle River — mouth of Purvis Creek	78	0.219	83
Turtle River — upper range	90	0.276	70
East River	85	0.249	68
Gibson Creek	88	0.245	73
Jointer Creek (reference)	85	0.255	62
Clubbs Creek (reference)	78	0.229	77
Laboratory control water	88	0.248	87

a. Mean weight of surviving mysids.

Sediment tests using Japanese medaka embryos

Survival, lesions, and hatching time were the endpoints evaluated in sediment toxicity tests using embryos of Japanese medaka (*Oryzias latipes*) conducted by Sprenger et al. (1997). The medaka embryos were placed in rearing solutions containing an LCP site sediment or reference site sediment. Sediments were collected from four sites along the LCP ditch in the Purvis Creek marsh, with Location 1 nearest Purvis Creek and Location 4 adjacent to the outfall pond. Survival was determined at three days post-hatch. Either 10 or 11 embryos were exposed to each sediment in single exposures (i.e., no replicates were used).

The results of the tests are shown in Table 5.3. Despite the differences in Aroclor 1268 and mercury sediment concentrations, medaka survival was similar (and not statistically significantly different) in each site sample compared to the reference sample. Lesions were observed in medaka in all treatments, and the number of lesions ranged from one in medaka exposed to reference sediment to seven in medaka exposed to Location 2 sediment. The study authors did not conduct statistical tests on the lesion data. The major lesions observed were heart edema, low blood flow, tail abnormalities, and yolk sphere hemorrhage. Sprenger et al. (1997) note that the observed lesions are consistent with lesions associated with PCBs, dioxins, and furans, and could also be consistent with mercury exposure.

Table 5.3. Results of medaka embryo toxicity tests by Sprenger et al. (1997).

Parameter	Location 1	Location 2	Location 3	Location 4	Reference
Total mercury (mg/kg dw)	34.0	15.0	90.0	170.0	0.1
Aroclor 1268 (mg/kg dw)	2.3	56.0	70.0	150.0	0.1
Medaka toxicity test survival (%)	89.9	90	100	89.9	100
Medaka lesions (number observed)	6	7	2	6	1

Sediment tests using amphipods and shrimp

Sprenger et al. (1997) conducted toxicity tests in which the marine amphipod *Leptocheirus plumulosus* and brown shrimp (*Penaeus vannamei*) were exposed to sediment from the LCP site and from a reference site (results are also presented in Horne et al., 1999). Amphipods and shrimp were exposed to site sediments collected from the same five sites used by Sprenger et al. (1997) for medaka embryo toxicity tests. Standard toxicity test methods were used, with exposure periods of 10 and 14 days for amphipods and shrimp, respectively. Test endpoints were survival and sediment avoidance for both organisms, and behavioral observations for amphipods.

The results of the Sprenger et al. (1997) study are shown in Table 5.4. Sediment from Location 4 (the LCP ditch draining the site outfall) was the most contaminated, with a mercury concentration of 170.0 mg/kg (dw) and an Aroclor 1268 concentration of 150.0 mg/kg (dw). Although amphipod survival was lowest at Location 4 (63%), the survival rate was not statistically different from that observed in reference sediment (78%). Shrimp survival also was not significantly different across sites, and no behavioral abnormalities were reported. Furthermore, mortality and sediment contaminant concentrations were not correlated across the samples, suggesting that the observed variability in survival was not caused by the contaminants measured.

Table 5.4. Results of shrimp and amphipod toxicity tests using site sediment by Sprenger et al. (1997).[a]

Parameter	Location 1	Location 2	Location 3	Location 4	Reference
Total mercury (mg/kg dw)	34.0	15.0	90.0	170.0	0.1
Aroclor 1268 (mg/kg dw)	2.3	56.0	70.0	150.0	0.1
Amphipod toxicity test survival (%)	92	83	68	63	78
Shrimp toxicity test survival (%)	-	100	-	97	94

a. Locations are the same as those used by Sprenger et al. (1997) in the Japanese medaka tests shown in Table 5.3.

A dash indicates that the test was not performed at that location.

PTI and CDR (1998) also conducted sediment toxicity tests with *Leptocheirus plumulosus*. Sediments were collected from 40 locations within the Purvis Creek marsh in areas planned for U.S. EPA's sediment removal action, and from two reference sites. Endpoints evaluated were survival and growth.

Mean survival in amphipods ranged from 63% to 100% in site sediment samples and from 87% to 98% in reference area sediment samples. Compared to one of the reference areas, amphipod survival was reduced in 3 of the 40 LCP site samples. Compared to the other reference area, survival was reduced in 5 of the 40 LCP site samples. Growth was reduced in amphipods exposed to 5 of the 40 LCP site stations compared to one reference area, and 1 of 40 compared to the other reference area. Overall, the reduced survival and growth effects observed were not correlated with sediment concentrations of Aroclor 1268, mercury, or lead.

Sediment and porewater tests using amphipods

Winger et al. (1993) evaluated the toxicity of sediments and porewater collected from Purvis Creek to the amphipod *Hyalella azteca*. The most potentially toxic reach of the creek was first identified by collecting sediments from the length of the creek and using them in Microtox assays, which measure the effect of porewater extracted from sediments on the bioluminescence of the bacterium *Photobacterium phosphoreum*. Two Purvis Creek locations near the mouth of the LCP ditch (Stations 6 and 7) were identified as the most toxic and used in the amphipod tests. Sediment and porewater from a Florida site were used as reference. Juvenile (2-3 mm) *H. azteca* were exposed to either sediment or porewater extracted from the sediment using 10-day static toxicity tests. Endpoints evaluated were mortality and leaf consumption.

The results of the toxicity tests are shown in Table 5.5. In the sediment exposures, no mortality was observed in any of the treatments. However, 56% and 76% of *H. azteca* exposed to the two LCP site porewaters died, compared to 5% mortality in reference porewater (these differences were statistically significant). Feeding rates were statistically significantly reduced in all site exposures (sediment and porewater) compared to reference exposures. The study authors attribute the porewater mortality and reduced feeding rates to toxicity from PCBs and/or mercury.

Table 5.5. Results of sediment and porewater amphipod tests by Winger et al. (1993).

Parameter	Station 6	Station 7	Reference
Sediment PCBs (μg/g)	95.1	67.3	0.04
Sediment mercury[a] (μg/g)	17.8	24.7	0.16
Sediment exposure mortality (%)	0	0	0
Sediment exposure feeding rate (mg/day)	48[b]	57[b]	73
Porewater exposure mortality (%)	76[b]	56[b]	5
Porewater exposure feeding rate (mg/day)	26[b]	44[b]	59

a. Includes methylmercury.
b. Reported by the study authors as statistically significantly different from reference.

5.2.2 Field bioaccumulation and toxicity studies using oysters

Caged oyster study

In a field bioaccumulation study designed to monitor uptake of chemicals, survival, growth, and general condition of oysters exposed to site surface water, oysters were deployed at nine study sites in the Purvis Creek estuary and two reference sites (PTI and CDR, 1998). Although the authors describe using both native and hatchery oysters, it appears that only hatchery oysters were deployed. Three different oyster deployments were conducted: the first was a 90-day exposure of oysters placed at locations throughout the Purvis Creek and Turtle River, the second was a 78-day exposure of oysters placed in the LCP ditch only, and the third was a 70-day exposure of oysters placed at single locations in Purvis Creek and the Turtle River (as well as at two reference sites).

The results of the study are given in Table 5.6. Survival of the first batch of deployed oysters was poor across all stations, including at reference sites. The lowest survival rate was 1% in the upper Turtle River, and the highest survival rate 22% in the Purvis Creek headwaters. In the second deployment, survival was higher at the single station sampled (LCP ditch), at 26%. In the third deployment, survival at the Purvis Creek station was 19%, compared with 29% and 30% at the

Table 5.6. Results of caged oyster survival study by PTI and CDR (1998).

Location	Test 1 90-day survival (%)	Test 2 78-day survival (%)	Test 3 70-day survival (%)
LCP Ditch	10	26	
Middle Purvis Creek	4		19
Upper Purvis Creek	8		
Purvis Creek headwaters	22		
Lower Purvis Creek	14		
Turtle River — mouth of Purvis Creek	-[a]		-[a]
Turtle River — upper range	1		
East River	3		
Gibson Creek	2		
Jointer Creek (reference)	2		30
Clubbs Creek (reference)	13		29

a. Sample was lost.
A blank indicates no sample was deployed at that station.

two reference locations (the oysters deployed in the Turtle River were lost). PTI and CDR (1998) note that the overall low survival observed in the two batches was believed to be the result of temperature and salinity stress rather than contaminant exposure.

Resident and caged oyster study

Matta et al. (1998) conducted a study on caged and resident oysters in 1997. The purpose of the study was to investigate the survival, growth, and PCB and mercury bioaccumulation in oysters at the LCP site.

Caged oysters were deployed at eight Purvis Creek marsh locations and two reference stations for 62 days. In addition, resident oysters were collected from three of the eight locations: the LCP ditch, the mouth of the ditch, and the mouth of Purvis Creek. Resident oysters were collected one or two days before deployment of caged oysters from within 10 m of the deployment locations.

Upon recovery from the field, the caged oysters were depurated overnight. Individual oysters were measured for growth rate as the change in whole animal wet weight over the exposure period, and composites of three oysters were analyzed for concentrations of mercury, methylmercury, lead, and PCBs (as Aroclors) and for lipid and water content. Composites of resident oysters were analyzed for the same suite of analytes as the caged oysters.

Results of the study are given in Table 5.7. The tissue chemistry results show that oysters deployed at the site accumulated PCBs and mercury. No PCBs were detected in oysters deployed in reference areas, whereas oysters deployed at the site accumulated PCBs at concentrations up to 218 μg/kg (ww). Aroclor 1268 was the only PCB mixture detected in the deployed oysters. Similarly, mercury concentrations in site-deployed oysters were up to 604 μg/kg (ww), compared to 18 and 21 μg/kg (ww) at the reference sites. Concentrations of mercury, methylmercury, and Aroclor 1268 showed an overall pattern of decline with distance from the head of the LCP ditch.

In addition to differences in PCB and mercury uptake, several other significant differences were observed between caged oysters deployed at the site and those deployed at the two reference stations. Oysters at all Purvis Creek and Purvis Creek marsh stations exhibited significantly reduced growth compared to reference site oysters. End-of-test lipid content was significantly lower in oysters from five of eight site stations compared to the reference stations, and end-of-test water content was significantly higher at all site stations compared to the two reference stations.

Resident oysters, which presumably had been exposed to site conditions for a longer period than caged oysters, had lower lipid levels and higher water content than the caged oysters. Resident oysters averaged 0.6% lipids and 91% water content, whereas caged oysters after the experiment averaged 1.1% lipids and 85% water content. However, PCB concentrations in resident oysters were lower than in caged oysters, although the lower concentrations could be related to the lower lipid content in the resident oysters.

In summary, the findings of this study are that:

- Oysters in Purvis Creek and the Purvis Creek marsh are exposed to and bioaccumulate PCBs and mercury. The type of PCBs accumulated and the spatial pattern of accumulation are consistent with the LCP site being the dominant source of the contaminants.

- Caged oysters deployed at the site for 62 days had reduced growth rates, lower lipid content, and higher water content than oysters deployed at reference stations. In addition, resident site oysters (that have most likely been exposed for more than 62 days) had lower lipid content and higher water content than caged oysters deployed at the site. This information suggests that exposure to some environmental factor(s) in Purvis Creek and the Purvis Creek marsh results in adverse effects to oysters.

Together, this information suggests that exposure to PCBs and/or mercury causes adverse effects to oysters in Purvis Creek and the Purvis Creek marsh.

Table 5.7. Results of oyster field deployment study by Matta et al. (1998) (mean ± SD).

Station	Location	Aroclor 1268 (µg/kg ww)	Hg (µg/kg ww)	Methyl Hg (µg/kg ww)	Pb (mg/kg ww)	Lipid content (%)	Water content (%)	Growth rate (mg/wk)
Pretest	Hatchery oysters	ND	29 ± 1.5	30.7	0.09 ± 0.02	1.0 ± 0	97.0 ± 5.2	NA
1	Reference	ND	21 ± 3.0	13.3	0.15 ± 0.03	1.4 ± 0.2	88.9 ± 9.8	695 ± 557
2	Reference	ND	18 ± 1.4	18.8	0.17 ± 0.07	1.3 ± 0.2	85.8 ± 12.9	938 ± 580
3	Tributary of ditch	54.3 ± 5.0	604 ± 27	492	0.20 ± 0.01	1.1 ± 0.1	86.5 ± 11.7	425 ± 332
4	Ditch near outfall	218.0 ± 225	357 ± 123	378	0.2 ± 0	0.4 ± 0.6	100 ± 0	296 ± 353
5	Mouth of ditch at Purvis Creek	49.5 ± 0.71	237 ± 12	396	0.18 ± 0	0.9 ± 0.0	100 ± 0	328 ± 328
6	Tributary of Purvis Creek	ND	579 ± 395	176	0.14 ± 0.01	0.9 ± 0.1	90.9 ± 7.9	263 ± 226
7	Tributary of Purvis Creek	ND	342 ± 49	176	0.16 ± 0.02	1.2 ± 0.1	89.7 ± 8.9	417 ± 280
8	Tributary of Purvis Creek	29.3 ± 34	329 ± 9.7	129	0.15 ± 0.01	0.9 ± 0.1	90.0 ± 8.6	284 ± 221
9	Mouth of tributaries at Purvis Creek	ND	147 ± 35	136	0.11 ± 0.01	0.9 ± 0.1	92.7 ± 6.4	615 ± 368
10	Mouth of Purvis Creek at Turtle River	17.2 ± 6.6	186 ± 7.9	179	0.17 ± 0.01	1.1 ± 0.2	89.4 ± 9.2	511 ± 392

ND = not detected.
NA = not applicable.

5.2.3 Benthic macroinvertebrate community studies

Sprenger et al. (1997) and Horne et al. (1999)

Sprenger et al. (1997) conducted a benthic invertebrate community study at the LCP site as part of U.S. EPA's ERA (results are also reported in Horne et al., 1999). The investigation consisted of macroinvertebrate community sampling at four sites in the Purvis Creek marsh and at one reference site, analysis of PCB and mercury concentrations in sediment and in epibenthic organisms (fiddler crabs and periwinkle snails) from each survey site, and amphipod toxicity testing using sediments collected at each site (the amphipod tests were discussed in Section 5.2.1). The four sites in the Purvis Creek marsh were in the LCP ditch, with Location 1 nearest Purvis Creek and Location 4 adjacent to the outfall pond.

Total invertebrate density showed no relationship to PCB or mercury sediment concentration. However, there were trends in both community composition and functional feeder group composition (Table 5.8). Community composition, measured as the percentage of the total community consisting of six taxonomic groups, is shown in Figure 5.1. Horne et al. (1999) concluded that the benthic communities at the reference site and Location 1, the least contaminated site, were dominated by nematodes and oligochaetes, whereas benthic communities in Locations 2 through 4, which were moderately to highly contaminated areas, were dominated by polychaetes. Polychaete percent community composition was positively associated with mercury and PCB concentrations, whereas nematode percent community composition was negatively associated with PCB and mercury concentrations and total organic carbon (TOC). Oligochaete percent community composition was positively associated with TOC content and negatively associated with mercury concentration (Horne et al., 1999).

Results of the trophic level analysis are given in Figure 5.2. Contaminated areas were dominated by surface feeders, whereas uncontaminated areas had approximately even percentages of surface and subsurface feeders. The percentage of surface feeders was positively associated with mercury and PCB concentrations. The percentage of subsurface feeders was positively associated with increasing TOC in sediment. Although not a statistically significant relationship, the percentage of carnivores decreased with increasing contamination (Horne et al., 1999). Horne et al. (1999) conclude that the high proportion of polychaetes and low proportions of oligochaetes and nematodes at contaminated sites may reflect differences in the tolerance of these groups to environmental disturbance, including contaminants.

Although the results of amphipod toxicity testing (described in further detail in the preceding section) revealed no evidence of acute toxicity to the benthic community, Horne et al. (1999) conclude that chronic community-level effects and contaminant bioaccumulation are still of concern. The presence of high concentrations of PCBs and mercury in marsh sediments and

Table 5.8. Results of benthic macroinvertebrate study by Sprenger et al. (1997) and Horne et al. (1999).

Parameter	Location 1	Location 2	Location 3	Location 4	Reference
Sediment Aroclor 1268 (mg/kg dw)	2.3	56.0	70.0	150.0	0.1
Sediment mercury (mg/kg dw)	34.0	15.0	90.0	170.0	0.1
Total organic carbon (%) in sediment	4.16	1.27	1.73	0.78	3.61
Crustacea (% community composition)	0.30	0.85	1.63	0.54	2.57
Gastropod (% community composition)	0	0	0.18	0	1.54
Insect (% community composition)	0.37	0.24	0	0.18	5.14
Nematode (% community composition)	51.07	1.57	5.98	3.42	12.34
Oligochaete (% community composition)	25.98	12.19	36.05	18.72	45.50
Polychaete (% community composition)	22.29	85.15	56.16	77.14	32.90
Carnivore (% trophic composition)	0.60	0.12	0	0.19	0.80
Subsurface feeder (% trophic composition)	63.25	16.75	51.92	22.60	56.30
Surface feeder (% trophic composition)	36.14	83.13	48.08	77.21	42.90
Aroclor 1268 conc. (mg/kg dw) in fiddler crab	4.9	40.0	43.0	-	0.08
Mercury conc. (mg/kg dw) in fiddler crab	0.7	2.0	2.6	-	0.06
Aroclor 1268 conc. (mg/kg dw) in *Littorina*	-	4.2	-	-	0.05
Mercury conc. (mg/kg dw) in *Littorina*	-	33.1	-	-	0.6

A dash indicates that the parameter was not evaluated at that location.

Source: Horne et al., 1999.

fauna, and the correlation of PCB and mercury concentrations in fiddler crabs and *Littorina* with sediment concentrations (Table 5.8), indicate that benthic invertebrates accumulate contaminants from sediment and transfer them to higher trophic levels (Horne et al., 1999).

PTI and CDR (1998)

PTI and CDR (1998) evaluated benthic macroinvertebrate assemblages at two stations in the Purvis Creek marsh and two reference stations located near Jointer Creek. The results of the study are presented in Table 5.9. PTI and CDR (1998) did not analyze their results statistically because of small sample size; however, they concluded that total number of taxa, total number of individuals, and mean density of individuals in sediment collected are similar between LCP and reference samples. PTI and CDR (1998) did not present an analysis of possible community or trophic level alterations in benthic macroinvertebrate assemblages in the Purvis Creek marsh similar to that conducted by Horne et al. (1999).

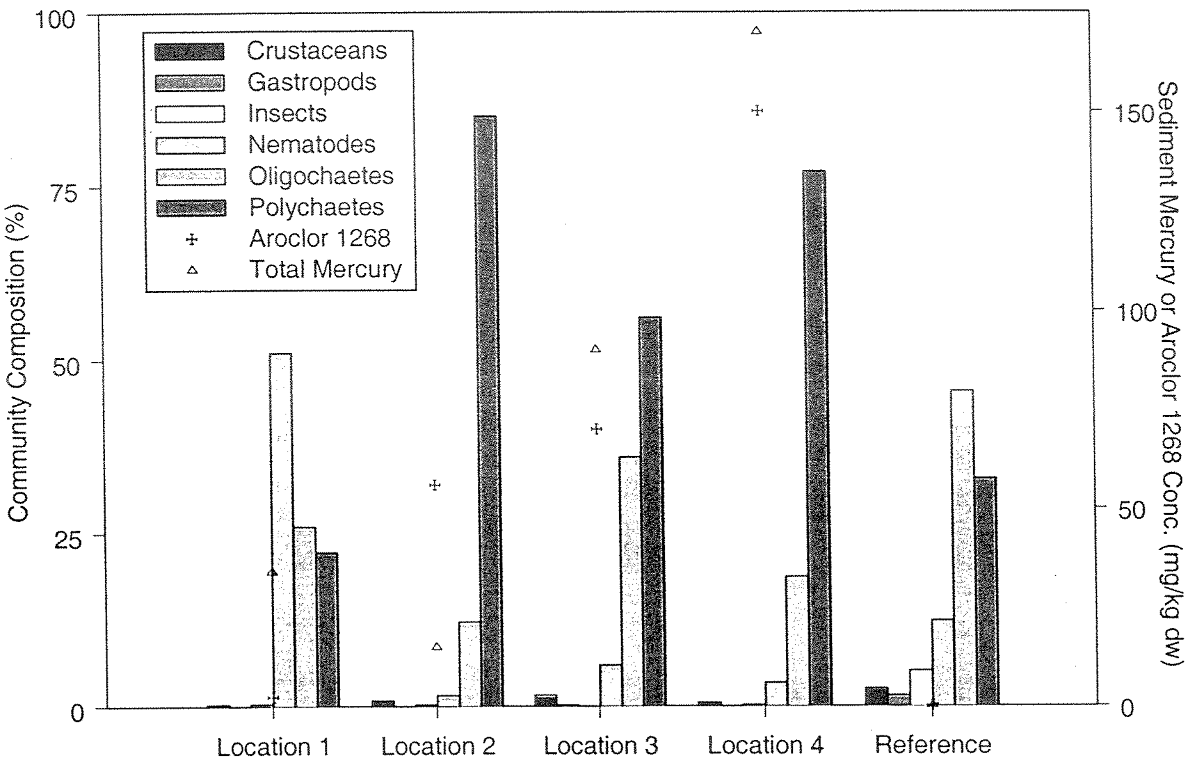

Figure 5.1. Results of the Horne et al. (1999) community composition analysis for the benthic macroinvertebrate community in the Purvis Creek estuary.

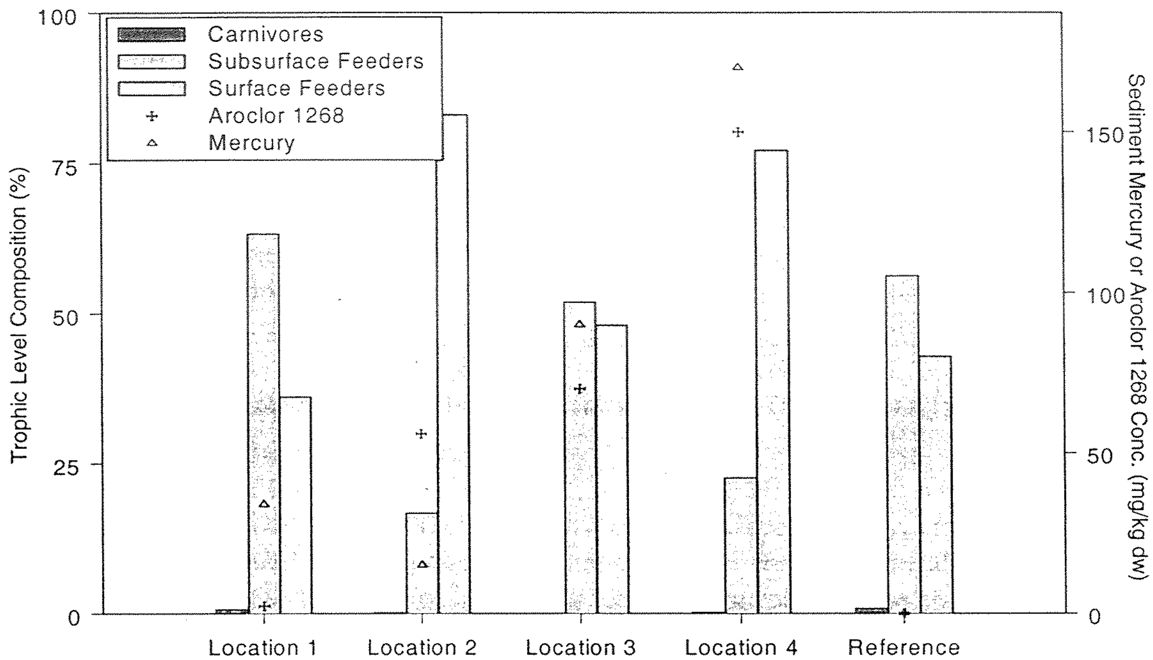

Figure 5.2. Results of the Horne et al. (1999) trophic level analysis for the benthic macroinvertebrate community in the Purvis Creek estuary.

Table 5.9. Results of benthic macroinvertebrate study by PTI and CDR (1998).

Sampling location	Total no. of taxa	Total no. of individuals	Mean density (no. individuals/m^2)
LCP, midchannel	7	47	2,089
LCP, channel bank	7	19	844
Reference, midchannel	9	30	1,333
Reference, channel bank	5	18	800

Table 5.10. Summary of site toxicity and adverse effects studies.

Study	Exposure media	Organism(s) tested	Exposure duration	Results
PTI and CDR (1998)	Site surface water	Mysids, sheepshead minnows	7-9 days	No effects on survival, growth, fecundity, teratogenicity
	Site sediment	Amphipods	"Chronic"	Reduced survival, growth at from 1 to 5 of 40 Purvis Creek station
	Site sediment and surface water (field deployment)	Oysters	70-90 days	Test failed because of high mortality in all treatments
Sprenger et al. (1997)	Site sediment	Japanese medaka	Through egg development and hatching	Increased incidence of lesions
		Amphipod and brown shrimp	10-14 days	No effects on survival or sediment avoidance behavior
Winger et al. (1993)	Site sediment	Amphipod	10 days	No effects on survival reduced feeding activity
	Site porewater (extracted from sediment)	Amphipod	10 days	Reduced survival and feeding rate
Matta et al. (1998)	Site sediment and surface water (field deployment)	Oysters	62 days	Decreased lipid content, increased water content
Horne et al. (1999)	Field study	Benthic macroinvertebrate community	NA	Shift in community to polychaetes and surface feeders, associated with PCBs and mercury

Sediment concentrations causing effects to benthic macroinvertebrates

PCBs

To evaluate the potential for sediment PCBs to cause toxicity to benthic macroinvertebrates, we use the "consensus" sediment effects concentrations estimated by MacDonald Environmental Services (1999) for the Hudson River NRDA. MacDonald compiled and reviewed existing publications in which sediment effect concentrations were derived from co-occurring data on PCB sediment concentrations and adverse effects to benthic organisms. The different sediment effect concentrations considered by MacDonald are those used by NOAA (effects range-low and effects range-medium), Ontario Ministry of the Environment (lowest effect level and severe

Table 5.11. Receptors, exposure routes, and site contaminant data used in the evaluation of potential adverse effects.

Receptor	Route of exposure evaluated	Type of site data evaluated
Benthic macroinvertebrates	Direct contact with sediment	Concentrations of PCBs, mercury, PAH in sediment
	Direct contact with surface water	Concentrations of mercury in surface water
Fish	Direct contact with surface water	Concentrations of mercury in surface water
	Dietary exposure	Concentrations of PCBs in benthic invertebrates
	All exposure routes	Concentrations of mercury in fish tissue
Birds	Dietary exposure	Concentrations of PCBs and mercury in fish

effect level), Environment Canada (minimal effect threshold and toxic effect threshold), Florida Department of Environmental Protection (threshold effects level and probable effects level), and Washington State for Puget Sound (apparent effects thresholds).

MacDonald estimated the "consensus" sediment effects concentrations by first separating each threshold into either freshwater or marine/estuarine, and then categorizing the thresholds into one of three levels: threshold effect concentrations (TEC), midrange effect concentrations (MEC), and extreme effect concentrations (EEC) (Table 5.12). Each increasing level from TEC to EEC represents an increased probability of observing adverse effects. For the purposes of this preliminary evaluation document, we use the marine/estuarine MEC to EEC range of 0.47 to 1.7 mg/kg (dw) as a potential toxic thresholds range for PCB concentrations in sediment.

Table 5.12. MacDonald Environmental Services (1999) consensus sediment effects levels for PCB toxicity to benthic invertebrates.

Effect level	Sediment concentration (mg/kg dw)
Threshold effect concentration (TEC)	0.048
Midrange effect concentration (MEC)	0.47
Extreme effect concentration (EEC)	1.7

Mercury, Lead, and PAHs

The Florida Department of Environmental Protection (Florida DEP) sediment quality assessment guidelines (SQAGs) for Florida coastal waters (MacDonald, 1994) were used to evaluate the potential for sediment mercury and PAH concentrations at the site to cause adverse effects to benthic macroinvertebrates. MacDonald (1994) developed the SQAGs using a weight of evidence approach applied to a database of effect concentrations (i.e., low observed effect concentrations) and no-effect concentrations (i.e., no observed effect concentrations) to aquatic organisms. MacDonald (1994) estimated two effects concentrations, a threshold effect level (TEL), the sediment contaminant concentration below which adverse effects are unlikely, and a probable effect level (PEL), the sediment contaminant concentration above which adverse effects are likely. The TEL and PEL for mercury, lead, and PAHs are shown in Table 5.13. Given that these sediment effects levels were selected for coastal waters in the State of Florida, they may also be applied to coastal waters of Georgia, such as those at the LCP site. For this preliminary evaluation document, we use the TEL to PEL range for each contaminant as a potential toxic effects range.

Table 5.13. MacDonald (1994) sediment effects levels mercury, lead, and PAH toxicity to benthic invertebrates.

Effect level	Mercury (mg/kg dw)	Lead (mg/kg dw)	PAHs (mg/kg dw)
Threshold effect level (TEL)	0.13	30.2	1.7
Probable effect level (PEL)	0.70	112	16.8

Surface water concentrations causing effects to benthic macroinvertebrates and fish

Adverse effects concentrations for direct toxicity from exposure to surface water are estimated only for mercury, since available site data on PCB concentrations in surface water are limited in their usefulness because of high detection limits (see Chapter 4), and because data on PAH and lead concentrations in site surface waters are very limited.

To evaluate the potential for measured mercury concentrations in site surface water to cause direct toxicity to aquatic biota, we use the U.S. EPA AWQC for the protection of saltwater aquatic life. An extensive database is available on the toxicity of mercury to saltwater aquatic organisms, and the U.S. EPA used this database in developing their AWQC. The value of 0.94 µg/L is compared to measurements of dissolved mercury in site surface water.

Dietary exposure concentrations causing effects to fish

Adverse effects concentrations for fish dietary exposure were estimated for PCBs based on a literature survey and review. These estimated dietary effects levels are compared to PCB concentrations measured in invertebrates at the site. Very few published studies are available on the dietary toxicity of mercury or PAHs to fish, so these contaminants were not evaluated for this exposure route.

The literature review identified studies in which fish toxicity was observed in fish fed dietary doses of PCBs. These studies were then reviewed and screened to eliminate studies in which:

- the dietary doses were not quantified, or cannot be determined from the information provided in the study
- no adverse effects were observed.

The studies are listed in Table 5.14. Dietary effects concentrations reported in the literature range from 1 mg/kg (dw), which caused increased thyroid activity in coho salmon in the study by Mayer et al. (1977), to 2,500 mg/kg (dw), which caused 50% mortality to minnows in the study by Bengtsson (1980). The four studies showing adverse effects at concentrations at or less than 5 mg/kg (dw) dietary dose all observed sublethal biochemical effects, whereas at dietary effects concentrations of 25 mg/kg (dw) and greater, effects were either decreased reproduction or mortality. More than half of the studies listed in Table 5.14 used salmonid species, and the toxicity of PCB mixtures to many fish species has not been studied.

One issue in applying the results of the studies listed in Table 5.14 to the LCP site is that most of the laboratory studies used Aroclor 1254, whereas Aroclor 1268 is the dominant PCB mixture that was released into the Purvis Creek marsh. Because of the high degree of chlorination of Aroclor 1268, the toxic potency of Aroclor 1268 is low when calculated on the basis of Ah-receptor mediated toxicity (i.e., dioxin-like toxicity; Kannan et al., 1998). In fact, Sawyer et al. (1984) found that Aroclor 1268 was 39% as potent as Aroclor 1254 in inducing ethoxyresorufin-O-deethylase (EROD) and 57% as potent in inducing aryl hydrocarbon hydroxylase (AHH), both of which are Ah-receptor mediated effects.

However, Ah-receptor mediated toxicity is but one type of toxicity that can be caused by PCBs (Safe, 1994). Few toxicity studies have been conducted using Aroclor 1268 for any types of organisms. A study by Kholkute et al. (1994) found that Aroclor 1268 had the same potency as Aroclor 1254 in reducing fertilization success, increasing oocyte degeneration, and increasing the incidence of abnormal mice embryos. A study on the dietary toxicity of PCB mixtures to chickens found that 20 mg/kg of Aroclor 1254 in the diet reduced the hatching success of fertile

Table 5.14. Toxicity of PCBs to fish based on dietary exposure.

Species	Lifestage	PCB type	Dietary dose (mg/kg dw)	Effect	Reference
Coho salmon		Aroclor 1254	1	Increased thyroid activity	Mayer et al., 1977
Rainbow trout		Aroclor 1254	1.9	Liver histology	Nestel and Budd, 1975
Rainbow trout		Aroclor 1254	3	Reduced vitellogenin	Chen et al., 1986
Atlantic croaker	Adult	Aroclor 1254	5	Reproductive indicators	Thomas, 1989
Cod	Adult	Aroclor 1254	25	Reduced reproduction	Sangalang et al., 1981
Cod	Adult	Aroclor 1254	125	Reduced survival — 90%	Sangalang et al., 1981
Rainbow trout	Juvenile	Aroclor 1254	300	Reduced growth	Cleland et al., 1988
Coho salmon		Aroclor mix	500	Reduced growth	Leatherland and Sonstegard, 1978
Coho salmon		Aroclor 1254	960	Reduced survival — 100%	Mayer et al., 1977
Minnow (*Phoxinus phoxinus*)	Adult	Clophen A50	2500	Reduced survival — 50%	Bengtsson, 1980

eggs to 80.3% (significantly different from controls, at $P<0.01$), whereas the chickens exposed to the same dose of Aroclor 1268 had 89.3% egg hatching success (not significantly different from controls; Lillie et al., 1974). Based on these studies, which show no or slight difference in the toxicity of Aroclor 1268 compared to Aroclor 1254, we assume that the fish dietary toxicity studies that used Aroclor 1254 are directly applicable to evaluating the toxicity of Aroclor 1268 at the LCP site.

Therefore, the PCB dietary concentrations in Table 5.14 that have been shown to cause toxicity to fish are compared in the next section to LCP site data on PCB concentrations in benthic macroinvertebrates (a common fish food item). Based on the studies listed in Table 5.14, we use a toxic effects range of 1 to 5 mg/kg (dw) PCBs to evaluate potential sublethal toxicity to fish via dietary exposure, and an effects concentration of 25 mg/kg (dw) PCBs to evaluate potential reproductive toxicity via dietary exposure. These concentrations are converted to a wet weight basis using the mean percent solids content that has been measured in invertebrate samples from the LCP site (31.1%, analysis not shown). Using this conversion factor, the dietary effects concentrations become 0.3 to 1.6 mg/kg (ww) for sublethal toxicity and 7.8 mg/kg (ww) for reproductive toxicity.

Fish whole-body concentrations associated with toxicity to fish

Concentrations of contaminants measured in fish tissue can, in some cases, be used to predict toxicity to the fish using the "critical body residue" approach (Barron et al., in press). We used this approach to evaluate mercury concentrations measured in fish at and near the LCP site. Data on PAH concentrations in fish tissue are sparse, and fish exposure to PCBs is evaluated via the dietary pathway, the primary exposure route for PCBs in fish.

However, it has not yet been shown that toxic effects can be predicted by whole-body (or any other tissue) mercury concentrations. Studies to date suggest that methylmercury accumulation in fish may not be a valid predictor of a defined toxicity level. As with inorganic metals, the rate of accumulation seems to be much more important than the terminal body residue concentration (Phillips and Buhler, 1978; Niimi and Kissoon, 1994). In rainbow trout, mercury concentrations in kidney, liver, spleen, brain, and muscle were inversely correlated with exposure concentrations (Niimi and Kissoon, 1994). Therefore, the measured residues at death were inversely proportional to the exposure concentration. The gill was the only tissue that contained similar mercury concentrations at death for all exposures. However, it has not been shown clearly in any studies whether a critical body residue approach is valid for evaluating mercury toxicity in fish.

Despite the uncertainties in the approach, we used values obtained from the literature on whole-body concentrations of mercury in fish that are associated with adverse effects to evaluate measured concentrations of mercury in fish from the LCP site. Wiener and Spry (1996) conducted a literature review of critical body residue concentrations for mercury in fish, and concluded the following:

- In muscle tissue, mercury concentrations of approximately 6 to 20 mg/kg (ww) in adult fish have been associated with adverse effects in field studies, and concentrations of approximately 5 to 8 mg/kg (ww) (for adult walleye) and 10 to 20 mg/kg (ww) (for adult salmonids) have been associated with toxicity in laboratory studies.

- In whole bodies, mercury concentrations equal to or greater than approximately 5 to 10 mg/kg (ww) are associated with toxicity.

- Behavioral effects may occur at concentrations below approximately 5 mg/kg (ww).

After the review by Wiener and Spry (1996) was published, Fjeld et al. (1998) published a study on behavioral effects associated with mercury tissue concentrations in fish. They found that larvae with 0.27 mg/kg (ww) mercury in whole-body tissue exhibited altered feeding behavior three years after the mercury exposure ended. Using an adult-to-embryo conversion factor of 0.2 from McKim et al. (1976), Fjeld et al. (1998) concluded that this larvae concentration corresponds to a maternal muscle concentration of 1.35 mg/kg (ww). Similarly, Matta et al. (1999) found that whole-body mercury concentrations of 0.47 mg/kg (ww) and higher caused increased mortality in male mummichogs, which the study authors state was probably attributable to behavioral alterations.

In the Lavaca Bay NRDA, the trustees used several ranges of mercury concentrations in fish muscle tissue to evaluate the potential for different types of adverse effects to be occurring (Texas General Land Office et al., 2000). The values used in the Lavaca Bay NRDA are shown in Table 5.15. For the purposes of this report, we use the same values as shown in Table 5.15 to interpret the mercury concentrations measured in fish tissue from the LCP site.

Table 5.15. Adverse effects ranges for mercury in fish muscle tissue used in the Lavaca Bay NRDA.

Type of adverse effects	Muscle tissue concentration range (mg/kg ww)
Behavioral and possible reproductive effects	0.5-1.0
Behavioral, probable reproductive, and early life stage effects	1.0-2.0
Probable chronic and sublethal effects	2.0-3.0
Chronic and sublethal effects	3.0-6.0
Lethal toxic effects	6.0-20.0

From: Texas General Land Office et al., 2000.

Dietary concentrations causing toxicity to birds

Birds that feed in the Purvis Creek marsh are exposed to PCBs and mercury that have accumulated in the tissues of fish and shellfish. The potential for dietary exposure to PCBs and mercury to cause toxicity to birds that feed in the marsh was evaluated by comparing PCB and mercury concentrations measured in fish and shellfish collected from the site to concentrations shown to cause toxicity to birds in laboratory dosing studies. This section describes dietary toxicity effects concentrations for birds exposed to PCBs or mercury.

PCBs

In their review of the dietary toxicity of PCBs to birds, Barron et al. (1995) concluded that dietary PCB concentrations of approximately 10-20 mg/kg (dw) are associated with reduced egg hatching success or other forms of embryomortality in bird species that are sensitive to PCBs, such as chicken. In a separate review of PCB toxicity to birds, Hoffman (1995) identified toxicity reference values for species such as terns, cormorants, and eagles that are approximately five times higher than toxicity reference values for chicken. This factor of five difference in the sensitivity of chickens versus other species to PCBs is consistent with the spread of acute LD50s reported by Heath et al. (1972). Thus, the dietary toxicity range of 10-20 mg/kg (dw) for chickens can be converted to a range of approximately 50-100 mg/kg (dw) for other, less sensitive species. Based on an assumed moisture content of 74% in whole body fish (Connolly et al., 1992), this range converts to 13-26 mg/kg on a wet weight basis. We use this range to evaluate the potential for PCBs in LCP site fish to cause reproductive impairment in birds that consume the fish.

PCBs have been shown to cause sublethal effects to birds at dietary concentrations less than those associated with reproductive impairment. Some of the most sensitive endpoints are sublethal effects to progeny of maternal birds fed PCBs (Barron et al., 1995). Table 5.16 lists the results of studies that have found sublethal effects in the progeny of birds fed PCBs. Thresholds for observing sublethal effects in the progeny of birds fed PCB-contaminated food range from 2.2 to 22 mg/kg (dw). These effects were observed in five studies using chickens and one using American kestrel. Effects observed included reduced growth rates, increased abnormalities (primarily edema and skeletal deformities), reduced shell thickness, and nutritional deficiencies. Based on the studies listed in Table 5.16, we use a range of 2 to 10 mg/kg (dw) in diet as indicative of causing adverse sublethal effects to progeny. Based on an assumed moisture content of 74% in whole body fish (Connolly et al., 1992), this estimated effects range translates to 0.5 to 2.6 mg/kg on a wet weight basis. Again, because the available data do not indicate that Aroclor 1268 is substantially less toxic via dietary exposure than Aroclor 1254, the effects range obtained from the studies listed in Table 5.16 can be applied directly to the data from the LCP site.

Mercury

Mercury is a persistent neurotoxin that bioaccumulates in aquatic systems. As with PCBs, higher trophic level organisms such as fish-eating birds typically have relatively high dietary exposure to mercury. Mercury can adversely affect survival, reproduction, behavior, growth and development, metabolism, histology, motor coordination, and blood and serum chemistry in birds (Eisler, 1987).

As part of the Great Lakes Water Quality Initiative, the U.S. EPA compiled and reviewed studies on the dietary toxicity of mercury to birds (U.S. EPA, 1995). Table 5.17 presents the mercury dietary toxicity studies used by U.S. EPA to derive protective levels of mercury exposure to birds in the Great Lakes. The dietary effects concentrations range from 0.5 mg/kg (ww) for reproductive impairment in mallard ducks to 65 mg/kg (ww) for mortality in goshawk. Based on the studies listed in Table 5.17, we used a possible threshold effects range of 0.5 to 5.0 (ww) for evaluating the potential for mercury accumulated in fish and shellfish to cause dietary toxicity to birds.

Table 5.16. Effects of adult avian exposure to PCBs on offspring sublethal endpoints.

Species	Endpoint life stage	Study duration (days)	Experiment duration (days)	PCB type	Lowest effect conc. (mg/kg dw diet)[a]	No effect conc. (mg/kg dw diet)[a]	Endpoint	Citation
Chicken	Chick	112	63	Aroclor 1248	2.2		Reduced growth rate in offspring	Lillie et al., 1974
Chicken	Chick	112	63	Aroclor 1254	2.2		Reduced growth rate in offspring	Lillie et al., 1974
American kestrel	Egg	180	180	Aroclor 1248	9[b]		Increased shell length and width	Lowe and Stendell, 1991
American kestrel	Egg	180	180	Aroclor 1248	9[b]		Reduced shell weight, thickness index, thickness	Lowe and Stendell, 1991
Chicken	Chick	14	14	Aroclor 1254	11		Increased vitamin E-selenium deficiency in progeny	Combs et al., 1975
Chicken	Chick	112	56	Aroclor 1242	11	5.5	Reduced growth rate in offspring	Lillie et al., 1975
Chicken	Chick	112	56	Aroclor 1248	11	5.5	Reduced growth rate in offspring	Lillie et al., 1975
Chicken	Embryo	112	63	Aroclor 1232	22		Increased number of embryo abnormalities	Cecil et al., 1974
Chicken	Embryo	112	63	Aroclor 1242	22	2.2	Increased number of embryo abnormalities	Cecil et al., 1974
Chicken	Embryo	112	63	Aroclor 1248	22	2.2	Increased number of embryo abnormalities	Cecil et al., 1974
Chicken	Embryo	112	63	Aroclor 1254	22	2.2	Increased number of embryo abnormalities	Cecil et al., 1974
Chicken	Chick	112	63	Aroclor 1232	22	2.2	Reduced growth rate in offspring	Lillie et al., 1974
Chicken	Chick	112	63	Aroclor 1242	22	2.2	Reduced growth rate in offspring	Lillie et al., 1974

a. Except where noted, wet or dry basis of dose was not specified. Reported dose was assumed to be on a wet weight basis and was converted to a dry weight basis assuming 10% moisture in commercial feed (Peakall and Peakall, 1973).
b. Dose was expressed in the study as wet weight in cockerel breast (3 mg/kg). Dose was converted to a dry weight basis assuming 66% moisture in cockerel breast (K.S. Kim et al., 1984; H.T. Kim et al., 1985).

Table 5.17. Toxicity of mercury to birds based on dietary exposure.

Species	Lifestage	Dietary dose (mg/kg)	Effect	Reference
Mallard ducks	Adult	0.5	Reproductive impairment	Heinz, 1974, 1975, 1976a, 1976b, 1979
Mallard ducks	Adult	3	Offspring mortality	Heinz and Loche, 1976
Japanese quail	Adult	4	Offspring mortality	Eskeland and Nafstad, 1978
Zebra finches	Unknown	5	Behavioral effects	Scheuhammer, 1988
Red-tailed hawks	1 year	5.2	Mortality	Fimreite and Karstad, 1971
Leghorn cockerel chicks	2 weeks	6	Growth	Fimreite, 1970
White leghorn hens	Adult	10	Egg production, weight, fertility	Scott, 1977
Leghorn cockerel chicks	2 weeks	18	Mortality	Fimreite, 1970
Domestic hen		15	Mortality	Heinz, 1974
Goshawk		65	Mortality	Heinz, 1974

All as cited in U.S. EPA, 1995.

Summary of exposure concentrations causing adverse effects

Table 5.18 summarizes the toxic effect threshold concentrations used to evaluate the sediment, surface water, and biota tissue contaminant concentrations of PCBs, mercury, lead, and PAHs that have been measured at the LCP site.

5.3.2 Results of the comparison of LCP site contaminant concentrations to toxic effects concentrations

This section compares concentration data for the LCP site area with the toxic effects concentrations or concentration ranges described in the previous section. The sources of the site data used in this comparison are presented in Section 2.3.

Table 5.18. Toxic effect concentration ranges used to evaluate potential injuries.

Receptor	Exposure	Contaminant	Effect	Toxic effects range
Benthic macroinvertebrates	Direct contact with sediment	PCBs	Toxicity to invertebrates	0.47-1.7 mg/kg (dw) in sediment
		Mercury	Toxicity to invertebrates	0.13-0.70 mg/kg (dw) in sediment
		Lead	Toxicity to invertebrates	30.2-112 mg/kg (dw) in sediment
		PAHs	Toxicity to invertebrates	1.69-16.8 mg/kg (dw) in sediment
	Direct contact with surface water	Mercury	Toxicity to invertebrates	0.94 µg/L (dissolved) in surface water
Fish	Direct contact with surface water	Mercury	Toxicity to fish	0.94 µg/L (dissolved) in surface water
	Dietary exposure	PCBs	Sublethal effects	0.3-1.6 mg/kg (ww) in diet
			Reproductive effects	7.8 mg/kg (ww) in diet
	All exposure routes	Mercury	Varies from sublethal to lethal	0.5-20 mg/kg (ww) in fish tissue
Birds	Dietary exposure	PCBs	Sublethal effects	0.5-2.6 mg/kg (ww) in diet
			Embryomortality	13-26 mg/kg (ww) in diet
		Mercury	Reproductive impairment	0.5-5.0 mg/kg (ww) in diet

Sediment concentrations and toxicity to invertebrates

The sediment concentration ranges for adverse effects to invertebrates are compared to two different sets of sediment contaminant data: (1) a dataset that includes all samples collected from the entire marsh area, including samples collected prior to EPA's removal action from the area where the action took place; and (2) a dataset that does not include samples collected from the removal areas prior to the removal action, but does include confirmatory samples collected from the areas after removal. The first dataset represents conditions in the marsh prior to EPA's removal action, and the second dataset represents conditions after the removal action.

PCBs

Prior to the U.S. EPA removal action, PCB concentrations throughout the marsh, including in areas north of the berm, exceeded the concentration range for effects to benthic invertebrates (Figure 5.3). PCB concentrations in most marsh samples from the southwestern portion of the marsh are less than the effects concentration range, with the exception of several samples along tributaries to Purvis Creek that may drain the LCP site area. The removal action took place in the marsh in the immediate area of the LCP site, and thus concentrations post-removal remain above the effects concentration range throughout the marsh (Figure 5.4). Confirmatory samples collected after the removal action show that the PCB concentrations in the sediment that remains is generally less than the effects concentration range (compare Figure 5.3 to 5.4).

These data show that PCB concentrations from many areas of the Purvis Creek marsh exceed the effects concentration range for adverse effects to benthic macroinvertebrates, for conditions both pre- and post-removal action.

Mercury

Mercury concentrations in surficial sediments prior to U.S. EPA's removal action (Figure 5.5) and after the removal action (Figure 5.6) exceed the sediment effects concentration range at nearly all locations in the Purvis Creek marsh that have been sampled. Concentrations are less than the threshold range at a few locations in upper Purvis Creek, a few locations in the southwestern portion of the marsh, and in sediments remaining after the removal action.

These data indicate that mercury concentrations in most areas of Purvis Creek and the Purvis Creek marsh, both pre- and post-removal, exceed the effects concentration range for toxicity to benthic invertebrates.

Lead

Lead concentrations in surficial sediment exceeded the sediment effects concentration range in the area of the marsh that was removed by the EPA, and in an area along the eastern edge of the marsh just north of the berm (Figure 5.7). Lead concentrations in the marsh outside of the areas removed by the EPA are generally within the sediment effects range (for areas near the site) or below the range (for areas farther from the site). Figure 5.8 shows that after the EPA removal, several sediment samples in the area of the removal still exceed the effects range, as do the samples along the eastern edge of the marsh just north of the berm. Lead concentrations in other areas of the marsh are either within or below the effects range.

These data indicate that lead concentrations in some areas of the marsh nearest the site exceeded the effects concentrations range for toxicity to benthic invertebrates both prior to and after EPA's removal action. The spatial extent of exceedences is much less than that for PCBs or mercury.

Figure 5.3. Purvis Creek marsh sediment PCB concentrations, representing conditions prior to the U.S. EPA removal action, compared to the sediment effects concentration range for toxicity to benthic invertebrates (0.47 to 1.7 mg/kg dw). Sediment PCB concentrations were averaged over the top 10 cm of sediment.

Figure 5.4. Purvis Creek marsh sediment PCB concentrations, representing conditions after the U.S. EPA removal action, compared to the sediment effects concentration range for toxicity to benthic invertebrates (0.47 to 1.7 mg/kg dw). Sediment PCB concentrations were averaged over the top 10 cm of sediment.

PAHs

No surficial sediment concentrations of PAHs exceed the toxic effects range, either before (Figure 5.9) or after (Figure 5.10) the EPA removal action. Most concentrations are below the effects range, and only a few scattered samples fall within the range. These data indicate that PAH concentrations in the Purvis Creek marsh are not sufficient to cause toxicity to benthic invertebrates.

Surface water mercury concentrations and toxicity to invertebrates and fish

The data available for the site do not indicate that direct exposure to dissolved mercury in surface water is likely to cause toxicity to invertebrates or fish. Of the approximately 41 sample results available for dissolved mercury from Purvis Creek, the Purvis Creek marsh, or the Turtle River, the highest concentration measured is 0.135 μg/L, which was found in a sample of marsh water by Matta et al. (1998). This concentration is well below the direct toxicity effects level of 0.94 μg/L. However, two important caveats should be noted: (1) many surface water samples that have been collected at the site have been analyzed for total mercury only and not for dissolved mercury; and (2) some of the studies that analyzed dissolved mercury used relatively high detection limits, such as the Sprenger et al. (1997) study, which used a mercury detection limit of 0.2 μg/L.

Therefore, although the data currently available do not indicate a potential for direct mercury toxicity to aquatic biota, the available surface water data are limited in their applicability and usefulness for evaluating toxic effects via this exposure route.

Invertebrate PCB tissue concentrations and dietary toxicity to fish

Figure 5.11 compares the PCB concentrations measured in whole-body invertebrate tissue from the site area to the fish dietary effects concentrations of 0.3 to 1.6 mg/kg (ww) for sublethal effects and 7.8 mg/kg (ww) for reproductive effects. The invertebrate PCB concentration data are shown separately for samples from Purvis Creek, from the Purvis Creek marsh, and from reference areas (as identified in the original studies). The organisms included in the data shown in Figure 5.11 are periwinkle snail, blue crab, fiddler crab, insects, annelid worms, shrimp, and echinoderms. Although not all of these organisms may be consumed by fish, the data on PCB concentrations in the whole-bodies of these organisms can be used as a preliminary indicator of potential dietary exposure for predatory fish.

Figure 5.5. Purvis Creek marsh sediment mercury concentrations, representing conditions prior to the U.S. EPA removal action, compared to the sediment effects concentration range for toxicity to benthic invertebrates (0.13 to 0.70 mg/kg dw). Sediment mercury concentrations were averaged over the top 10 cm of sediment.

Figure 5.6. Purvis Creek marsh sediment mercury concentrations, representing conditions after the U.S. EPA removal action, compared to the sediment effects concentration range for toxicity to benthic invertebrates (0.13 to 0.70 mg/kg dw). Sediment mercury concentrations were averaged over the top 10 cm of sediment.

Figure 5.7. Purvis Creek marsh sediment lead concentrations, representing conditions prior to the U.S. EPA removal action, compared to the sediment effects concentration range for toxicity to benthic invertebrates (30.2 to 112 mg/kg dw). Sediment surface lead concentrations are shown.

Figure 5.8. Purvis Creek marsh sediment lead concentrations, representing conditions after the U.S. EPA removal action, compared to the sediment effects concentration range for toxicity to benthic invertebrates (30.2 to 112 mg/kg dw). Sediment surface lead concentrations are shown.

Figure 5.9. Purvis Creek marsh sediment PAH concentrations, representing conditions prior to the U.S. EPA removal action, compared to the sediment effects concentration range for toxicity to benthic invertebrates (1.68 to 16.8 mg/kg dw). Sediment PAH concentrations were averaged over the top 10 cm of sediment.

Figure 5.10. Purvis Creek marsh sediment PAH concentrations, representing conditions after the U.S. EPA removal action, compared to the sediment effects concentration range for toxicity to benthic invertebrates (1.68 to 16.8 mg/kg dw). Sediment PAH concentrations were averaged over the top 10 cm of sediment.

Figure 5.11 shows that PCB concentrations measured in most of the invertebrates from the Purvis Creek marsh are within or exceed the effects concentration range for causing sublethal effects to fish, and many fall within or exceed the effects concentration range for causing reproductive effects. Whole-body invertebrate concentrations from Purvis Creek are lower than those from the marsh, but many still fall within the sublethal effects range. Concentrations from reference areas are all lower than those from the site and are below both effects ranges.

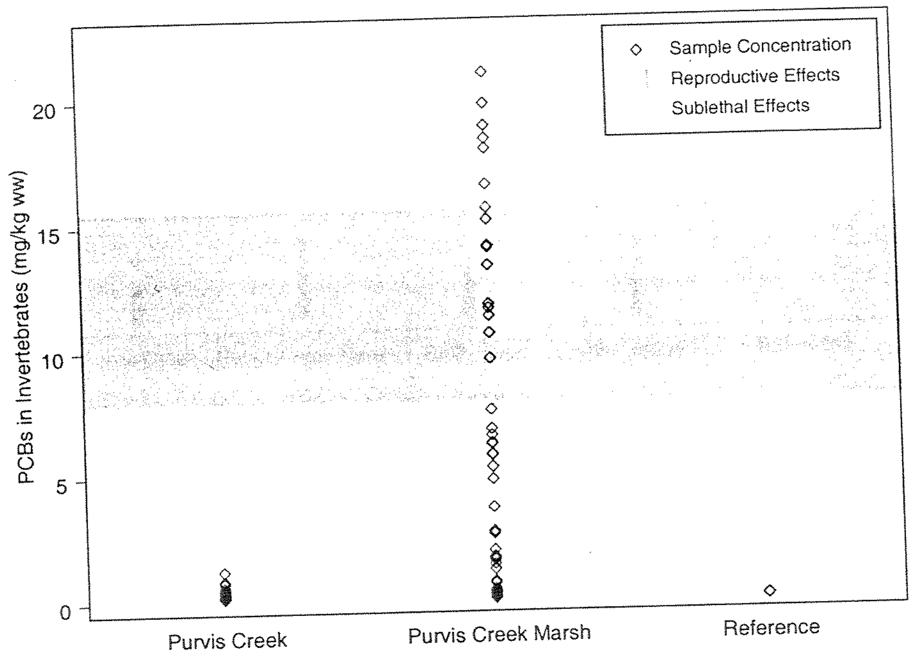

Figure 5.11. PCB concentrations measured in invertebrate whole bodies from the LCP site area compared to effects concentrations for dietary toxicity to fish.

These data show that PCB concentrations in invertebrates in the Purvis Creek marsh and in Purvis Creek may be sufficient to cause dietary toxicity to fish that consume invertebrates. Fish may suffer from sublethal effects and from adverse reproductive effects.

Fish mercury tissue concentrations and toxicity to fish

Figure 5.12 compares the mercury concentrations measured in fillet or whole-body fish tissue from the site area to the five effects concentration ranges for mercury toxicity to fish. The data are shown separately for samples from Purvis Creek, the Purvis Creek marsh, Turtle River, other areas within the estuary (e.g., South Brunswick River), and reference areas (as identified in the original studies).

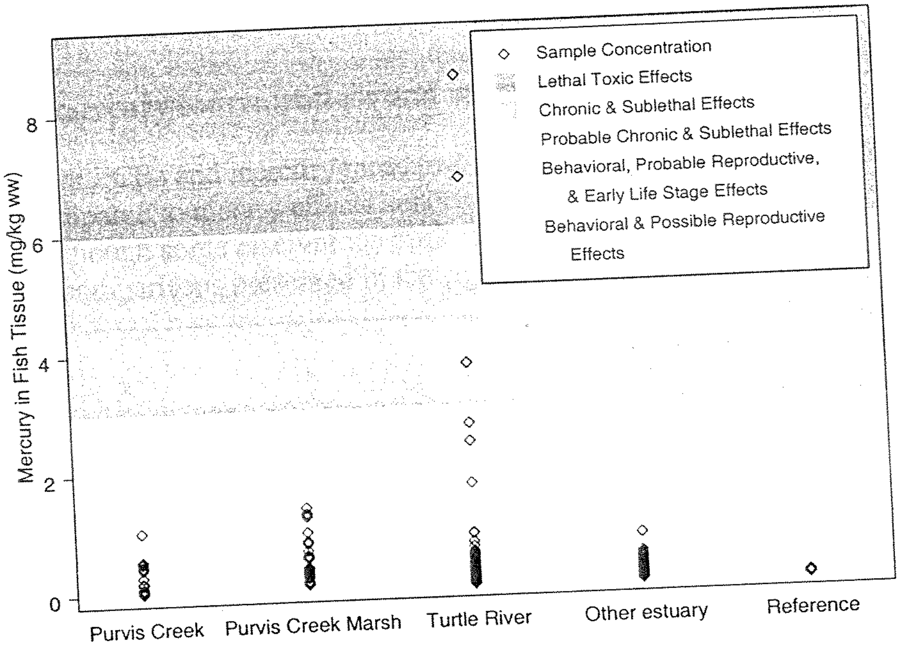

Figure 5.12. Mercury concentrations in fish whole bodies or fillets from the LCP site area compared to effects concentrations for toxicity to fish.

Mercury concentrations in fish tissue are most elevated in the Turtle River, where samples from two fish fall within the lethal effects range, one falls within the chronic and sublethal effects range, two fall within the probable chronic and sublethal effects range, and one falls within the behavioral, probable reproductive, and early life stage effects range. These higher concentration samples are all from the reach of the Turtle River into which Purvis Creek flows, similar to the spatial pattern of mercury in blue crabs that is described in Chapter 3. Mercury concentrations in some fish from Purvis Creek, the Purvis Creek marsh, and other areas within the estuary also fall within or exceed the effects ranges for behavioral and possible reproductive effects and behavioral, probable reproductive, and early life stage effects. Mercury concentrations in fish from reference areas are all well below the lowest toxic effects range.

Therefore, Figure 5.12 shows that fish in Purvis Creek, the Purvis Creek marsh, the Turtle River, and other areas within the estuary contain mercury concentrations that may be sufficient to cause behavioral, reproductive, and early life stage effects. In addition, some fish in the Turtle River near the LCP site contain concentrations that may be sufficient to cause lethality or chronic and sublethal effects.

Fish PCB and mercury concentrations and dietary toxicity to birds

Concentrations of PCBs and mercury measured in whole-body or fillet fish tissue from near the LCP site are compared to dietary effects concentrations for birds in Figures 5.13 and 5.14, respectively. Although some piscivorous birds may not feed exclusively in the areas shown in the figures, the comparisons presented in the figures provide a means of evaluating potential

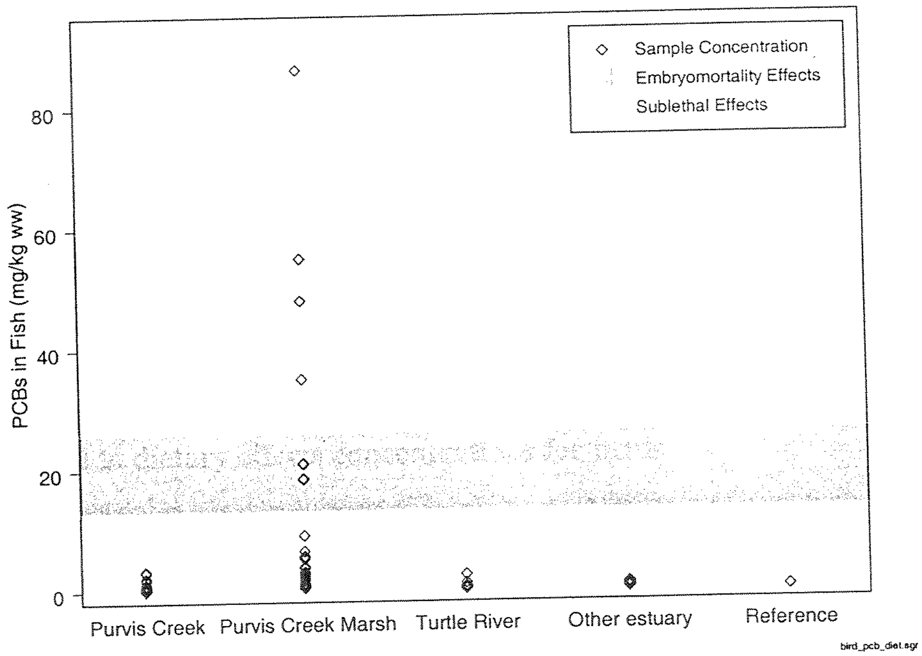

Figure 5.13. PCB concentrations in fish whole bodies or fillets from the LCP site area compared to dietary effects concentrations for birds.

levels of exposure for birds within each area.

Figure 5.13 shows that PCB concentrations in fish from Purvis Creek, the Purvis Creek marsh, the Turtle River, and other areas within the estuary fall within or exceed the effects concentration range for sublethal toxicity to birds that consume the fish. In addition, some samples from the Purvis Creek marsh fall within or exceed the dietary effects concentration range for causing embryomortality effects to birds. PCB concentrations in reference area fish are below both threshold ranges.

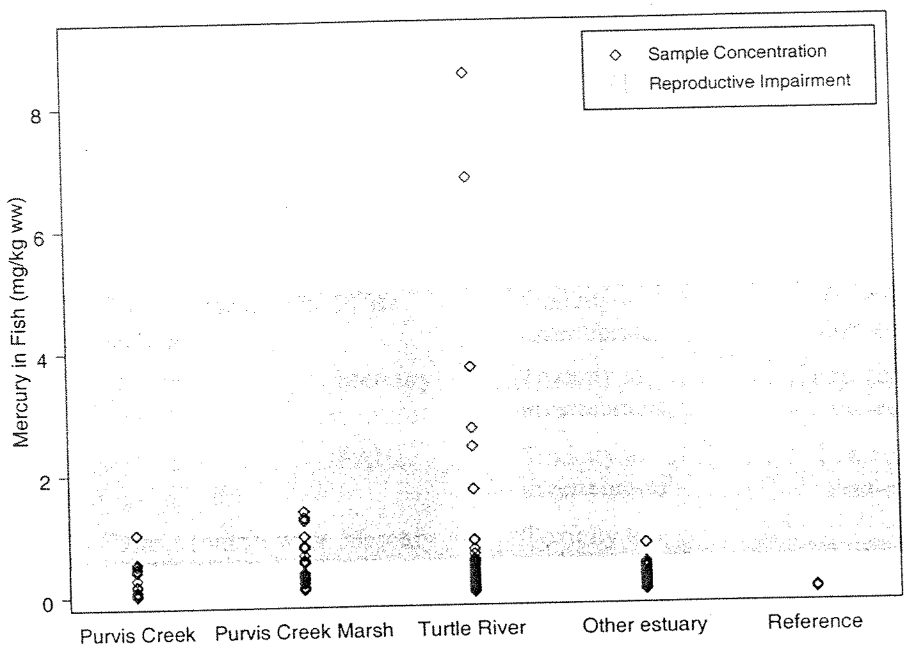

Figure 5.14. Mercury concentrations in fish whole bodies or fillets from the LCP site area compared to dietary effects concentrations for birds.

Mercury concentrations in some fish from areas near the LCP site also fall within or exceed the toxic effects concentration range for causing reproductive impairment in birds (Figure 5.14). The highest concentrations are in samples from the Turtle River near the LCP site, and these concentrations exceed the toxic effects range. Concentrations in some samples from each of the other areas near the LCP site fall within the toxic effects range, and all of the samples from reference areas are below the effects range.

These data show that PCB and mercury concentrations in fish from the LCP site area are sufficient to cause toxicity to birds that consume the fish. Toxic effects ranges for sublethal and embryomortality effects are exceeded by PCB concentrations in some fish, and the toxic effects range for reproductive impairment effects is exceeded by mercury concentrations in some fish. This analysis evaluated the exposure to each contaminant separately; simultaneous exposure to both site contaminants may lower the toxic effects ranges and result in increased toxicity to birds that consume fish from the LCP site area.

5.3.3 Summary

Table 5.19 presents a summary of the comparisons of site contaminant data to toxic effects ranges. For each comparison that was conducted, the table indicates whether at least some site data exceed the relevant toxic effects ranges or not.

Table 5.19. Summary of comparison of site concentration data to effects ranges.

Receptor	Exposure	Contaminant	Effect	Site data equal to or greater than effect range?
Benthic macroinvertebrates	Direct contact with sediment	PCBs	Toxicity to invertebrates	Pre-removal: Yes Post-removal: Yes
		Mercury	Toxicity to invertebrates	Pre-removal: Yes Post-removal: Yes
		PAHs	Toxicity to invertebrates	Pre-removal: No Post-removal: No
	Direct contact with surface water	Mercury	Toxicity to invertebrates	No
Fish	Direct contact with surface water	Mercury	Toxicity to fish	No
	Dietary exposure	PCBs	Sublethal effects	Yes
			Reproductive effects	Yes (Purvis Creek marsh only)
	All exposure routes	Mercury	Varies from sublethal to lethal	Yes
Birds	Dietary exposure	PCBs	Sublethal effects	Yes
			Embryomortality	Yes (Purvis Creek marsh only)
		Mercury	Reproductive impairment	Yes

Concentrations of PCBs and mercury that have been measured in site sediment and biota fall within or exceed effects ranges for toxicity to invertebrates, fish, and birds. Of the comparisons conducted, only dissolved mercury concentrations in surface water and PAH concentrations in sediment are below toxic effects ranges. The spatial extent of the potentially toxic concentrations varies with contaminant, receptor, and exposure type, as described in detail in the preceding section. In Purvis Creek and the Purvis Creek marsh, the following toxic effect concentrations are met or exceeded:

- sediment concentrations of PCBs and mercury sufficient to cause toxicity to benthic macroinvertebrates, both prior to and after the sediment removal action by the U.S. EPA
- invertebrate tissue concentrations of PCBs sufficient to cause sublethal and reproductive effects to fish that consume invertebrates
- fish tissue concentrations of mercury sufficient to cause behavioral, probable reproductive, and early life stage effects
- fish tissue concentrations of PCBs and mercury sufficient to cause sublethal effects and reproductive impairment in birds that consume the fish.

In the Turtle River, the following toxic effect concentrations are met or exceeded:

- invertebrate tissue concentrations of PCBs sufficient to cause sublethal effects to fish that consume invertebrates
- fish tissue concentrations of mercury sufficient to cause behavioral, probable reproductive, early life stage, and lethality effects
- fish tissue concentrations of PCBs and mercury sufficient to cause sublethal effects and reproductive impairment in birds that consume the fish.

The comparison of site contaminant data to effects concentrations should be interpreted with caution. There is uncertainty in the applicability of the toxic effects concentrations used in the comparison to the species, Aroclor type, contaminant bioavailability, contaminant exposure routes, and life history patterns specific to the LCP site. Also, the available site contaminant data are limited primarily to bulk sediment concentrations, and data on contaminant concentrations in surface water and biota tissues are fewer. Furthermore, we did not attempt to quantitatively address how exposure to multiple contaminants or other stressors could influence the toxicity of the contaminants at the site. Nevertheless, this preliminary comparison of available site contaminant data with toxic effect concentrations is a useful and informative means of assessing the potential for site contaminants measured in the environment to be causing toxicity to biota.

5.4 Conclusions

The preliminary evaluation of potential adverse effects injuries to biota was conducted by examining three pieces of evidence: the conclusions of the ecological risk assessments that have been conducted for the site; the results of site field and laboratory studies on the toxicity of site sediment and/or surface water to fish and invertebrates; and a comparison of site contaminant concentrations to effect concentrations for estimating toxicity to invertebrates, fish, and birds.

The two ecological risk assessments that have been conducted for the site arrived at different conclusions. The EPA risk assessment (Sprenger et al., 1997) concluded that the site poses risks to invertebrates, fish, mammals, and birds, whereas the risk assessment conducted by PTI and CDR (1998) concluded that the site poses no ecological risks. One key difference between the two assessments is that EPA's risk assessment was conducted before the removal action, whereas the PRP risk assessment was conducted to assess risks post-removal. The two assessments also used different methods and assumptions, which make their conclusions difficult to compare.

Most of the field and laboratory toxicity studies at the site have focused on acute exposure of invertebrates or fish to site sediment, water, or both. Overall, these studies do not provide any strong evidence that site sediment or surface water is acutely toxic to the organisms tested. However, two invertebrate toxicity studies that used longer exposure durations both found reduced growth or reduced survival. The results of the benthic macroinvertebrate community study, which found a shift in community and feeding group composition, are consistent with these results.

Lastly, PCBs and mercury released from the LCP site have accumulated in sediment and biota to concentrations that meet or exceed toxic effects concentrations. The comparison of site contaminant data with effects concentration ranges indicates that sediment PCB and mercury concentrations are sufficient to cause toxicity to invertebrates, invertebrate PCB concentrations are sufficient to cause dietary toxicity to fish, fish mercury concentrations meet or exceed concentrations associated with injury to fish, and fish PCB and mercury concentrations are sufficient to cause dietary toxicity to birds.

Overall, the available data indicate that adverse effect injuries may be occurring to exposed biological resources at the site.

6. Fish Consumption Guidelines, Fishing Closures, and FDA Exceedences

6.1 Introduction

This chapter summarizes information related to fish consumption guidelines, fishing closures, and exceedences of Food and Drug Administration (FDA) action and tolerance levels for fish and shellfish in Purvis Creek and the Turtle River. The information presented in this chapter can be used to evaluate injuries according to the following injury definitions in the DOI NRDA regulations:

- concentration of a released hazardous substance that is sufficient to exceed levels for which an appropriate State health agency has issued directives to limit or ban consumption of such organism [43 CFR § 11.62 (f)(1)(iii)], or
- concentration of a released hazardous substance that is sufficient to exceed action or tolerance levels established under Section 402 of the Food, Drug and Cosmetic Act, 21 U.S.C. 342, in edible portions of organisms [43 CFR § 11.62 (f)(1)(ii)].

6.2 Fish Consumption Guidelines and Fishing Closures

Section 6.2.1 describes the fish consumption guidelines and fishing closures issued by the State of Georgia for the Purvis Creek/Turtle River area as a result of PCB and mercury contamination in fish and shellfish. Section 6.2.2 briefly describes the basis for the guidelines, and Section 6.2.3 compares contaminant data for the site with the State's guideline trigger levels.

6.2.1 Description of the guidelines and closures

As a result of PCB and mercury contamination data collected in late 1991, the State of Georgia in 1992 issued a "precautionary advisory" against the consumption of seafood harvested in Purvis Creek and portions of Gibson Creek and the Turtle River (Georgia DNR, 1996). The precautionary advisory warned against the consumption of fish or shellfish from these areas specifically because of mercury and PCB contamination. The spatial extent of the advisory was based on the guideline areas shown in Figure 6.1. The 1992 advisory covered areas 1, 2, and 4 shown in Figure 6.1.

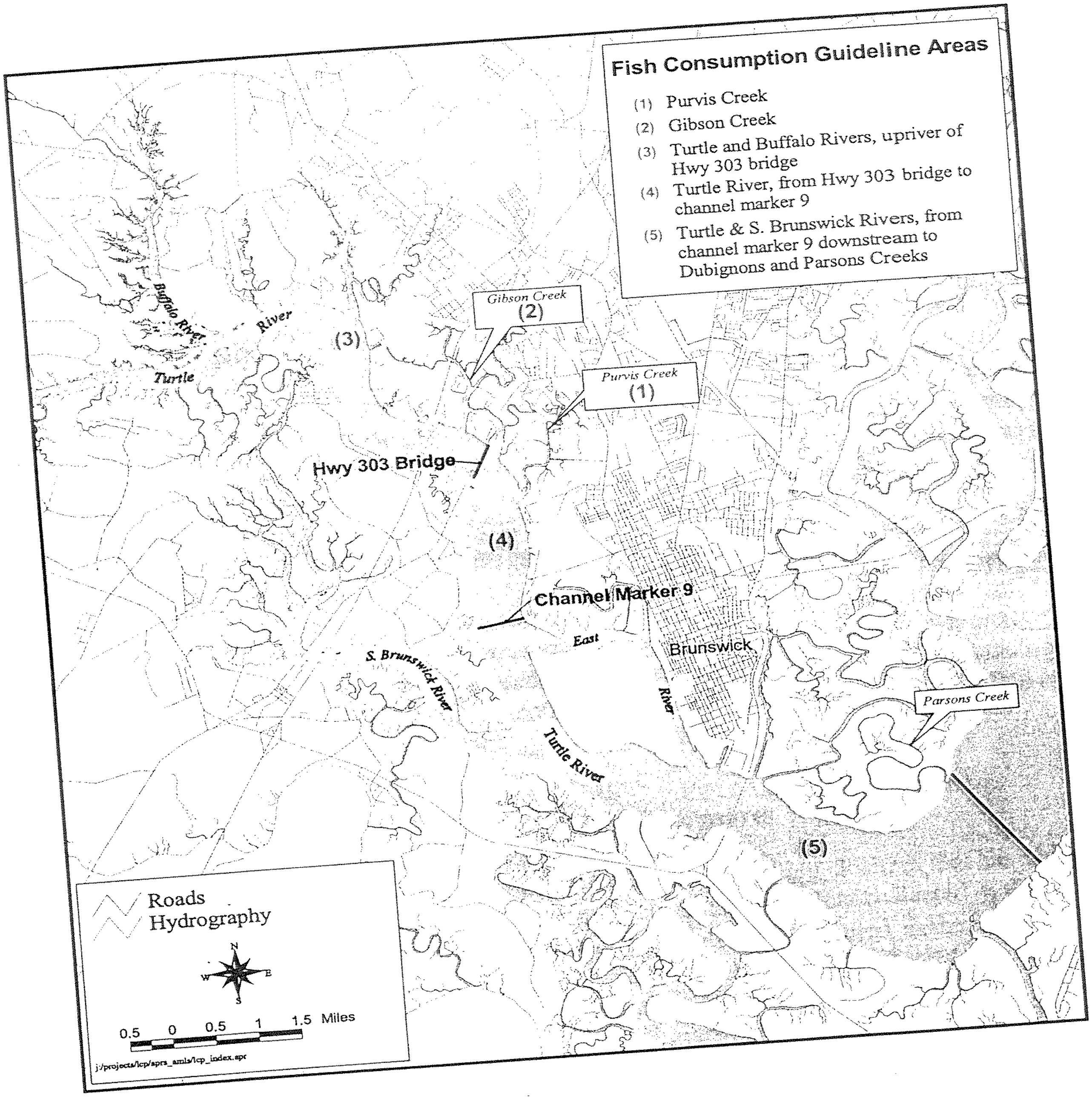

Figure 6.1. The fish consumption guideline areas in the Purvis Creek and Turtle River area.

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Figure 6.1. The fish consumption guideline areas in the Purvis Creek and Turtle River area.

In 1992 the Georgia DNR also closed the commercial fishery in the area. Purvis Creek and Gibson Creek (areas 1 and 2 in Figure 6.1) were closed to bait shrimp harvesting, and area 4 (the Turtle River near the site) was closed to commercial harvesting of any seafood (Georgia DNR, 1996). Several crabbers and bait shrimpers are believed to have harvested crab or shrimp in the closure area before the ban was issued. Commercial finfishing was most likely affected little by the ban since the commercial netting of finfish was not allowed in the closure area before the ban was issued (S. Shipman, Georgia Division of Coastal Resources, pers. comm., June 2000).

In 1993 the State issued a consumption advisory to replace the "precautionary advisory." The 1993 consumption advisory covered the same areas as the 1992 precautionary advisory; data collected in 1992 and 1993 did not indicate that there was a need to expand the area of the advisory (Georgia DNR, 1996).

A change made in the fish consumption advisory in 1995 reflected the State's revised approach to determining the need for and type of consumption guidelines specific to an area (see next section for a description). The new approach included lower trigger levels for issuing advisories, and it resulted in more restrictive and extensive "consumption guidelines" in the Purvis Creek/Turtle River area (the term was changed from advisories to guidelines in 1995). The guidelines established in 1995 have remained in place since then, and they are shown in Table 6.1. Mercury and PCBs are the contaminants responsible for the guidelines for all species and locations shown in Table 6.1 except for clams, mussels, and oysters in the Turtle River (guideline areas 3, 4, and 5 in Figure 6.1), consumption of which is banned by the National Shellfish Sanitation Program because of exceedences of fecal coliform standards (R. Manning, Georgia DNR, pers. comm., June 2000). The commercial fishing ban has remained in effect and unchanged since it was first issued in 1992.

6.2.2 Basis for the consumption guidelines and fishing closures

The fish consumption guidelines (or advisories) and commercial fishing closures that have been issued by the State have been based on PCB and mercury contaminant data collected in fish and shellfish in the area. Tests on the first samples, collected in late 1991, resulted in the issuance of the first precautionary advisory in 1992. Since then, the Georgia DNR has collected samples of fish and shellfish from the area for contaminants analysis in 1992, 1993, 1995, and 1997. Samples were of edible portions of organisms, and in all years but 1991 were analyzed as composites of individuals of the same species (Keller, 1998). The 1991 samples were analyzed as individual organisms (Georgia DNR, 1996). Table 6.2 summarizes the species and sample sizes collected in 1991, 1992, 1993, and 1995 (the 1997 data were not available for this report).

Table 6.1. Fish consumption guidelines from 1995 to present for Purvis Creek and the Turtle River.

Species	(1) Purvis Creek	(2) Gibson Creek	(3) Turtle and Buffalo rivers, upriver of Hwy 303 bridge	(4) Turtle River, from Hwy 303 bridge to channel marker 9	(5) Turtle and S. Brunswick rivers, from channel marker 9 downstream to Dubignons and Parsons creeks
All species[a]	Do not eat	Do not eat	—	—	—
Black drum	—	—	Do not eat	1 meal/month	1 meal/month
Blue crab	—	—	1 meal/week	1 meal/month	1 meal/week
Clams, mussels, and oysters	—	—	Do not eat[b]	Do not eat[b]	Do not eat[b]
Croaker	—	—	1 meal/month	1 meal/month	1 meal/week
Flounder	—	—	1 meal/week	1 meal/week	No restriction
Red drum	—	—	1 meal/week	1 meal/month	No restriction
Spotted sea trout	—	—	1 meal/month	1 meal/month	1 meal/month

a. Guidelines specifically apply to "blue crab, clams, mussels, oysters, shrimp, and other seafood."
b. Shellfish ban per the National Shellfish Sanitation Program.

Sources: Georgia DNR, 1998a, 1998b, 1999, 2000.

Table 6.2. Summary of fish and shellfish collections by Georgia DNR in Purvis Creek/Turtle River area.

Year	Species collected (common names)	Number of samples[a]
1991	Crab, oyster	10
1992	Black drum, blue crab, croaker, red drum, sheepshead, shrimp	10
1993	Black drum, blue crab, croaker, red drum, sheepshead, shrimp, spot, spotted seatrout, summer flounder	87
1995	Black drum, blue crab, croaker, flounder, mullet, red drum, sheepshead, shrimp, spot, spotted seatrout, yellow tail, whiting	162

a. In 1991, each sample was an individual organism. In all other years, samples were composites of five or more organisms.

To determine whether consumption guidelines should be issued for an area, the State compares the concentrations measured in the edible portions of organisms to trigger levels (described below). Consumption guidelines are issued when the mean concentration of at least three samples of a given species from a given location exceed the trigger level (R. Manning, Georgia DNR, pers. comm., June 2000). The exact nature of the advisory is determined based on the professional judgment of the Georgia DNR, in consultation with the Georgia Division of Coastal Resources (R. Manning, Georgia DNR, pers. comm., June 2000). Guidelines may be suggested for species for which no contaminant data are available, based on concentrations measured in similar species or in the same species in nearby areas. Guidelines may also be suggested for a species with a mean concentration below the trigger level if individual samples of the species were over the threshold (R. Manning, Georgia DNR, pers. comm., June 2000).

Before 1995, the trigger levels for PCB and mercury were the FDA levels of 2.0 mg/kg (ww) for PCBs and 1.0 mg/kg (ww) for mercury (Georgia DNR, 1996). In 1995, the State adopted new trigger levels that were based on a risk-based approach to evaluating human health effects from consumption of contaminated seafood (Georgia DNR, 1996). The risk-based trigger levels incorporate both cancer and noncancer endpoints for human health effects. In 1997, the trigger levels were again adjusted, this time because of a revision in the dose-response models used in the risk-based calculations (R. Manning, Georgia DNR, pers. comm., June 2000). Table 6.3 lists the different trigger levels for PCBs and mercury in fish and shellfish before 1995, from 1995 to 1997, and from 1997 to present. As the values in the table show, trigger levels for both PCBs and mercury dropped in 1995 when the State adopted a risk-based approach. In 1997, the revised trigger levels increased slightly for PCBs and decreased by a factor of approximately 3 for mercury. However, the advisories for the Purvis Creek/Turtle River area have not changed from those issued in 1995 (R. Manning, Georgia DNR, pers. comm., June 2000).

Table 6.3. Trigger levels used by Georgia DNR to issue fish consumption advisories/guidelines.

	PCBs (mg/kg ww)			Mercury (mg/kg ww)		
Advisory level	**Before 1995**	**1995-1997**	**Since 1997**	**Before 1995**	**1995-1997**	**Since 1997**
No restriction	<2.0	<0.07	<0.1	<1.0	<0.7	<0.23
One meal/week	NA	0.07-0.21	0.1-0.3	NA	0.7-2.1	0.23-0.70
One meal/month	NA	0.21-0.7	0.3-1.0	NA	2.1-7	0.70-2.3
Do not eat	NA	>0.7	>1.0	NA	>7	>2.3

6.2.3 Comparison of 1995 site data to Georgia trigger levels

This section presents a comparison of the State's trigger levels for issuing consumption guidelines to the 1995 data from the site. The 1995 data served as the basis for the consumption guidelines that were issued in 1995 and that have remained in place since. Before 1995, Georgia used the FDA action/tolerance levels as the consumption advisory trigger levels; the 1991, 1992, and 1993 Georgia DNR data are compared to the FDA action/tolerance levels in Section 6.3.

Figures 6.2 and 6.3 compare the 1995 concentrations of PCBs and mercury, respectively, in fish and shellfish from the Purvis Creek/Turtle River area to the 1995 Georgia DNR trigger levels. The figures show that 1995 PCB concentrations were higher than the levels that would trigger issuance of consumption guidelines. Specifically, PCB concentrations in 74% of Purvis Creek samples and 50% of samples from the other four guideline areas exceeded at least the lowest PCB trigger level required for issuance of an advisory. The "do not eat" threshold for PCBs was exceeded at least once in every guideline area. Mercury concentrations in 11% of Purvis Creek samples and 5.6% of samples from the other guideline areas exceeded at least the lowest mercury trigger level, and no samples exceeded the "do not eat" threshold for mercury.

In 1997, Georgia and the U.S. EPA conducted a joint sampling similar to the sampling in 1995. Based on the 1997 data, Georgia concluded that there were no overall changes in concentrations of contaminants and that, although some differences in frequency of action level exceedences were evident, fish and shellfish from the Purvis Creek/Turtle River system still had elevated contamination concentrations. Furthermore, the State concluded that any improvement from removal of contaminated soil and termination of PCB and mercury inputs had not yet become evident (Keller, 1998).

6.3 Exceedences of FDA Action and Tolerance Levels

According to DOI NRDA regulations, fishery resources can be injured if they contain concentrations of hazardous substances that exceed action or tolerance levels established under Section 402 of the Food, Drug and Cosmetic Act, 21 U.S.C. 342, in edible portions of organisms [43 CFR § 11.62 (f)(1)(ii)]. This section presents a comparison of PCB and mercury concentrations measured in edible portions of organisms collected from the Purvis Creek/Turtle River area to relevant FDA action or tolerance levels established by the act.

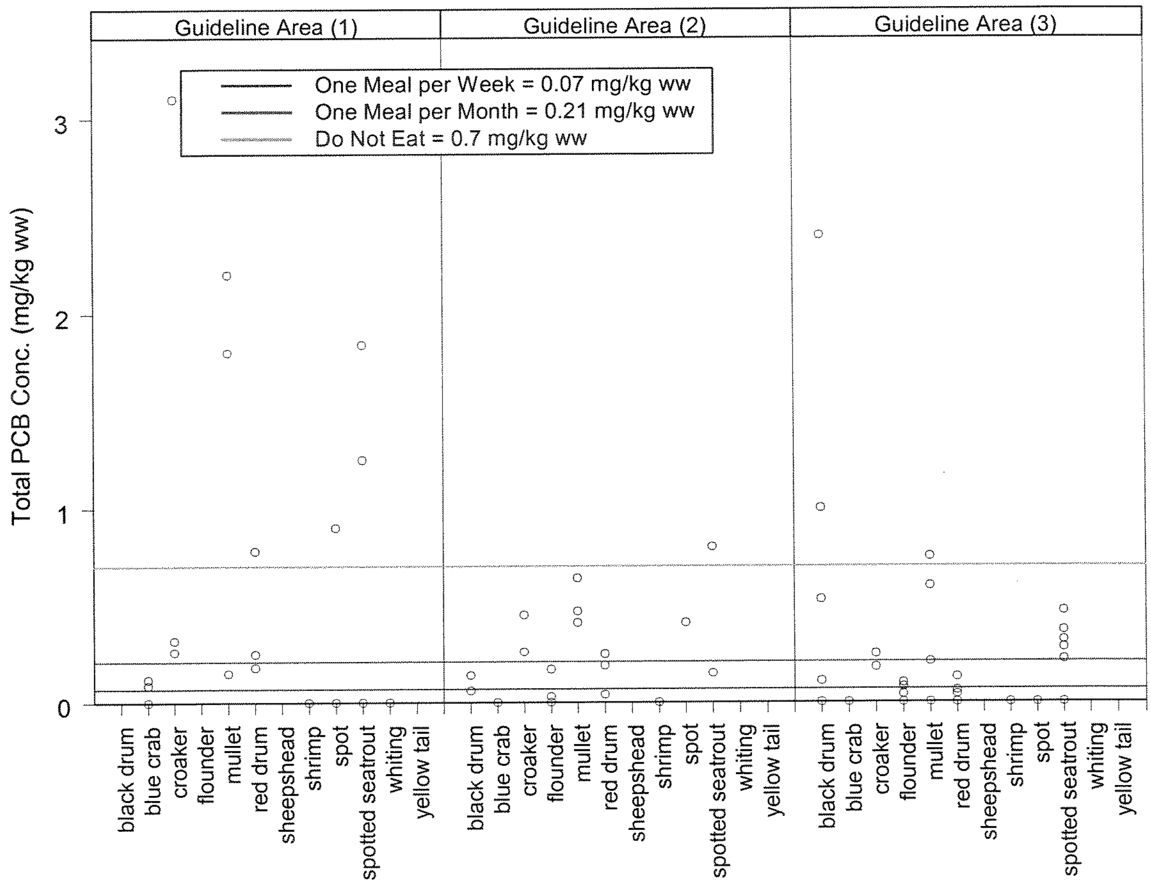

Figure 6.2. PCB concentrations in composite fish and shellfish samples collected in 1995 in comparison to the Georgia trigger levels for fish consumption guidelines. Guideline areas are shown in Figure 6.1. Samples in which PCBs were not detected are plotted at zero.

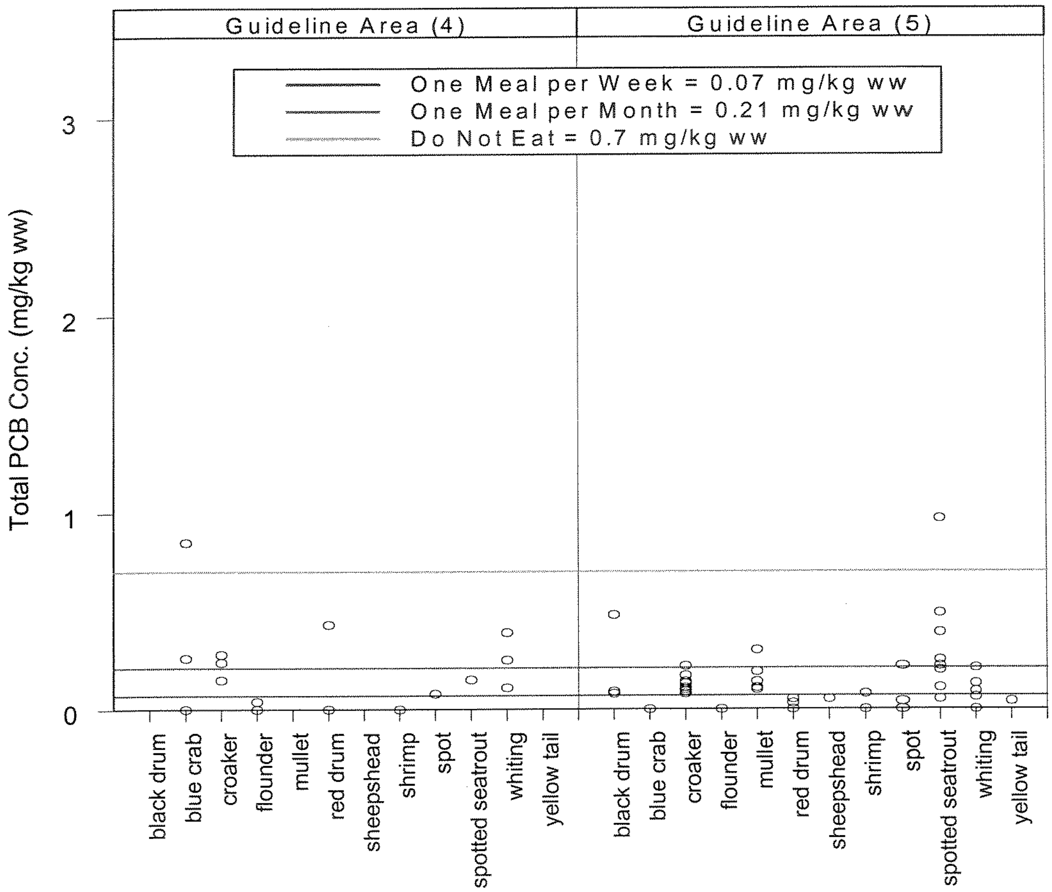

Figure 6.2 (cont.). PCB concentrations in composite fish and shellfish samples collected in 1995 in comparison to the Georgia trigger levels for fish consumption guidelines. Guideline areas are shown in Figure 6.1. Samples in which PCBs were not detected are plotted at zero.

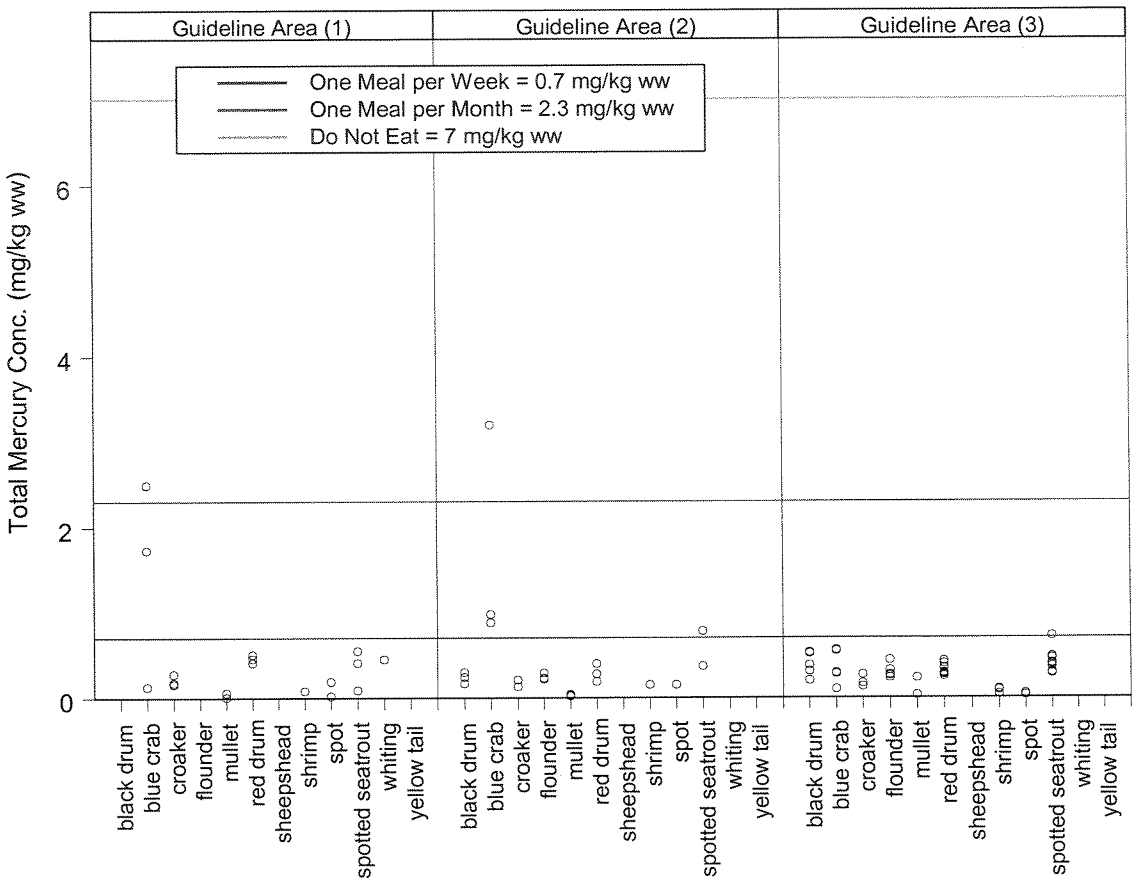

Figure 6.3. Mercury concentrations in composite fish and shellfish samples collected in 1995 in comparison to the Georgia trigger levels for fish consumption guidelines. Guideline areas are shown in Figure 6.1. Samples in which mercury was not detected are plotted at zero.

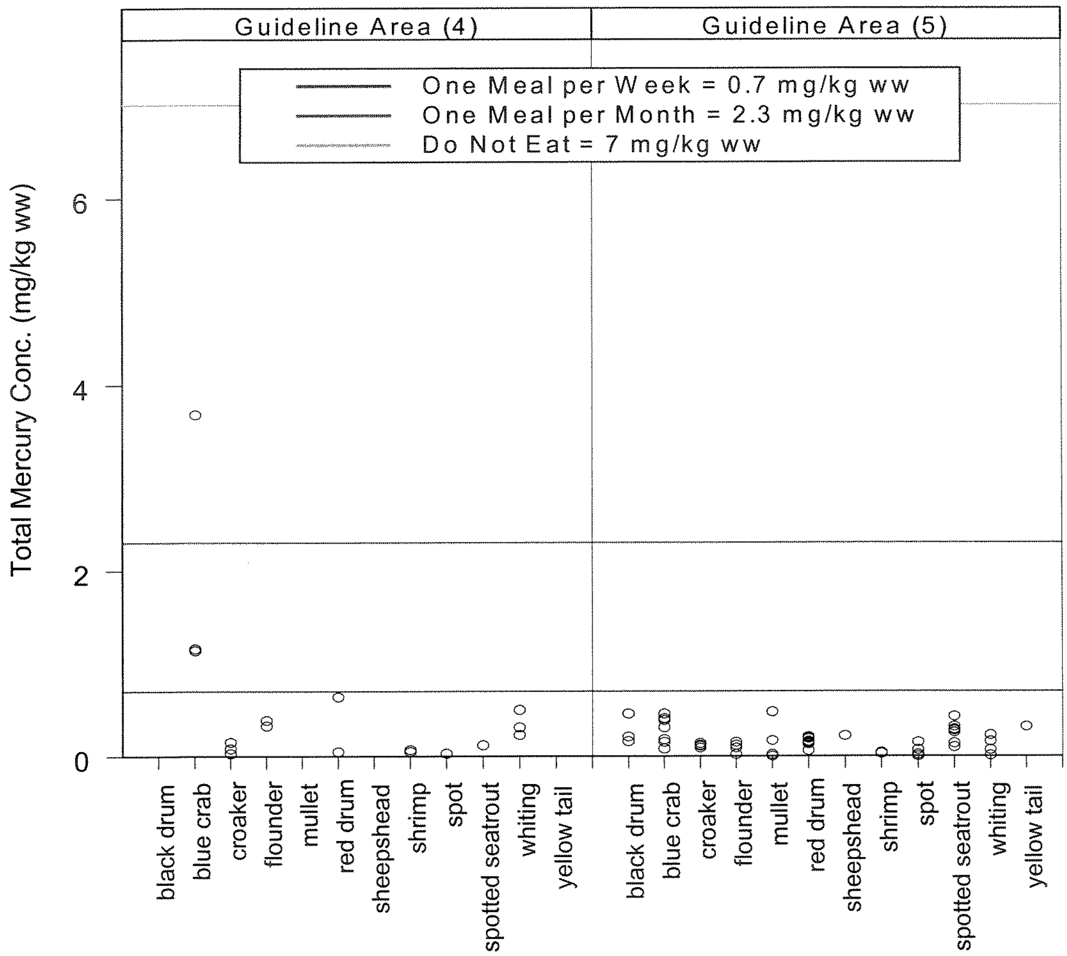

Figure 6.3 (cont.). Mercury concentrations in composite fish and shellfish samples collected in 1995 in comparison to the Georgia trigger levels for fish consumption guidelines. Guideline areas are shown in Figure 6.1. Samples in which mercury was not detected are plotted at zero.

The Food, Drug, and Cosmetic Act (21 U.S.C. 301 et seq.) authorizes the FDA to regulate food shipped in interstate commerce to protect public health. Sections 402 and 406 of the act prohibit food containing any added poisonous, deleterious, or unsafe substance from interstate commerce, unless the presence of the substance cannot be avoided. Section 406 enables the FDA to regulate levels of environmental contaminants that can enter food, and authorizes the FDA to limit the quantities of such substances by setting legal limits called tolerances or action levels.

The FDA threshold for PCBs is termed a "tolerance level," and the threshold for mercury is termed an "action level." The PCB tolerance level was 5 mg/kg from 1973 to 1984, has been 2 mg/kg since 1984 [38 FR 18096; 21 CFR 109.30 (a)(7)]. The action level for mercury has been 1 mg/kg since it was established in 1979 (FDA Administrative Guideline No. 7408.09).

6.3.1 Exceedences of FDA thresholds in Purvis Creek and the Turtle River

Figures 6.4 and 6.5 compare the concentrations of PCBs and mercury, respectively, measured in fish and shellfish from the Purvis Creek/Turtle River area to the relevant FDA level. The data shown include all fish and shellfish data collected by the Georgia DNR from 1991 through 1995.

Figure 6.4 shows that the PCB concentrations in the edible portions of some fish and shellfish collected from the area exceed the FDA tolerance level of 2 mg/kg (ww). Most of the exceedences occur in Purvis Creek, where 15% of the samples exceed the tolerance level. Outside of Purvis Creek (Turtle River and Gibson Creek), 0.41% of the samples exceed 2 mg/kg PCBs.

Figure 6.5 shows that the FDA action level for mercury is also exceeded in fish from the Purvis Creek/Turtle River area. Over all years of sampling shown, the mercury action level is exceeded in 15% of the samples from Purvis Creek and 6.5% of the samples from the Turtle River and Gibson Creek.

One feature of the site data should be noted here. For all years but 1991, the data points shown in Figures 6.4 and 6.5 are composites of five or more individual fish or shellfish. Therefore, the actual number of organisms in which the FDA tolerance or action levels is exceeded is greater than the number of data points in Figures 6.4 and 6.5 that are above the FDA levels. However, the percentage of individual organisms that exceed the FDA levels may or may not be similar to the percentage of the composite sample values that exceed the levels, depending on the nature of the compositing scheme and the underlying data distribution across individual organisms.

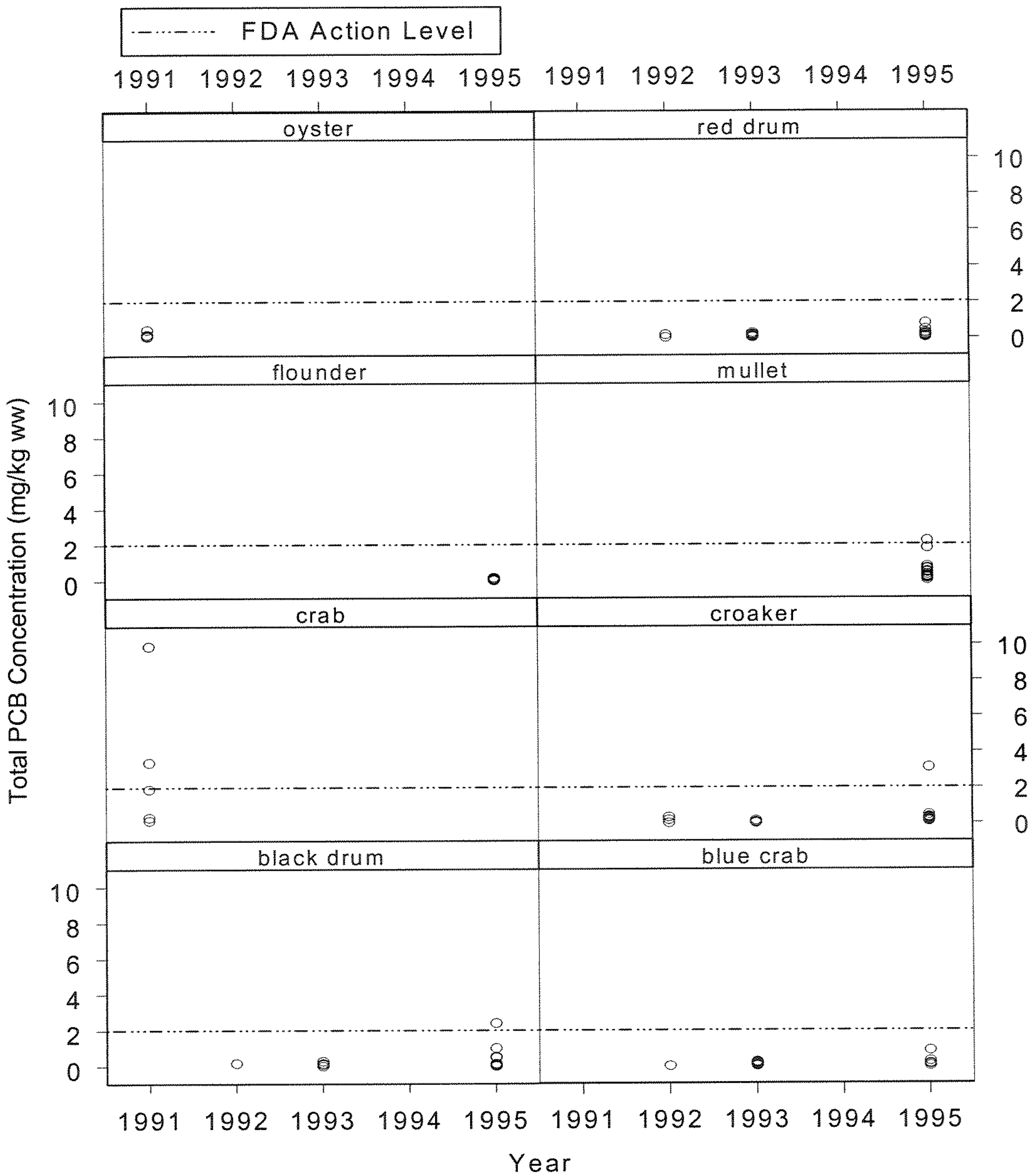

Figure 6.4. PCB concentrations in fish and shellfish collected from the Purvis Creek/Turtle River area compared to the FDA tolerance level of 2 mg/kg (ww). Samples in which PCBs were not detected are plotted at zero.

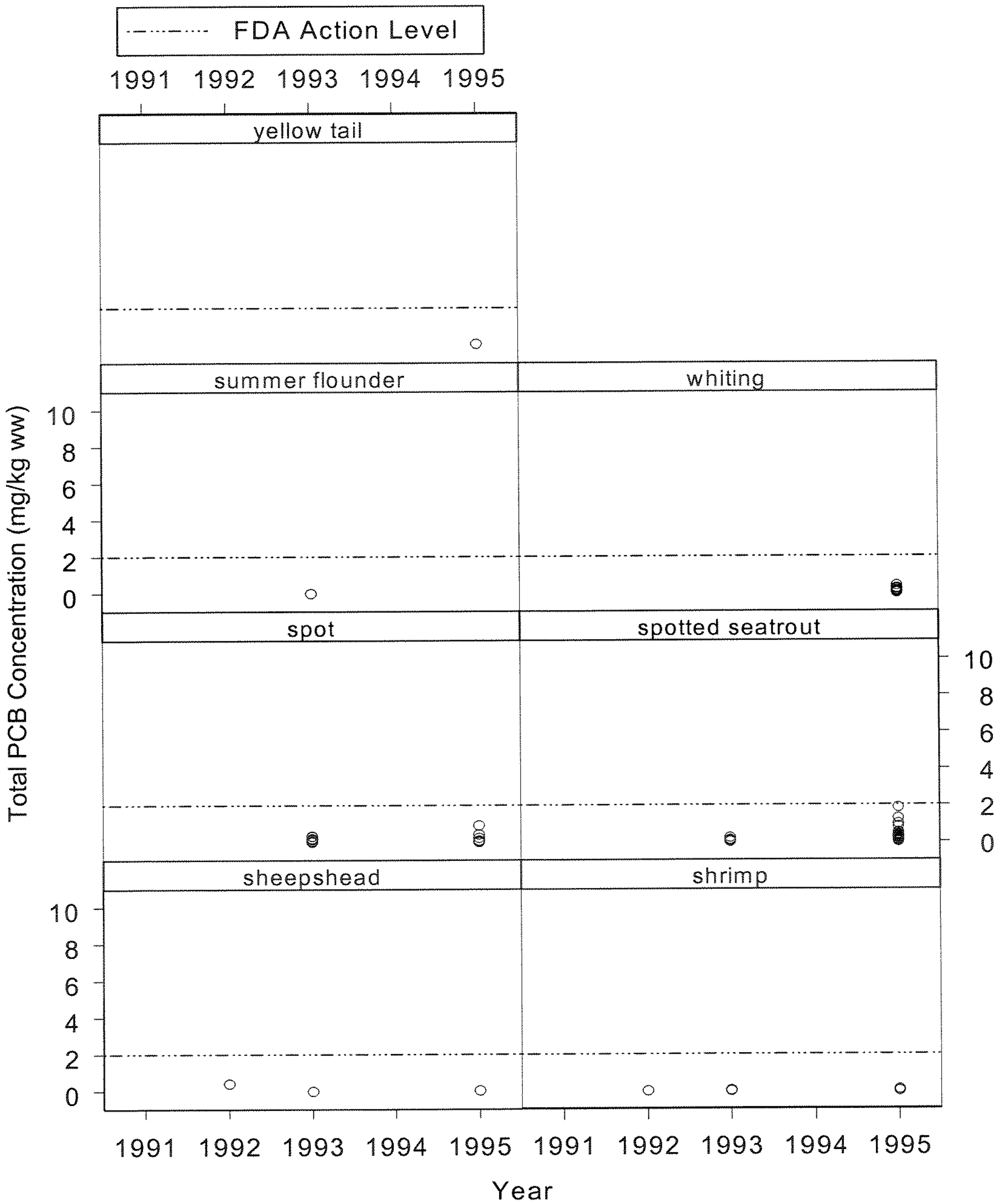

Figure 6.4 (cont.). PCB concentrations in fish and shellfish collected from the Purvis Creek/ Turtle River area compared to the FDA tolerance level of 2 mg/kg (ww). Samples in which PCBs were not detected are plotted at zero.

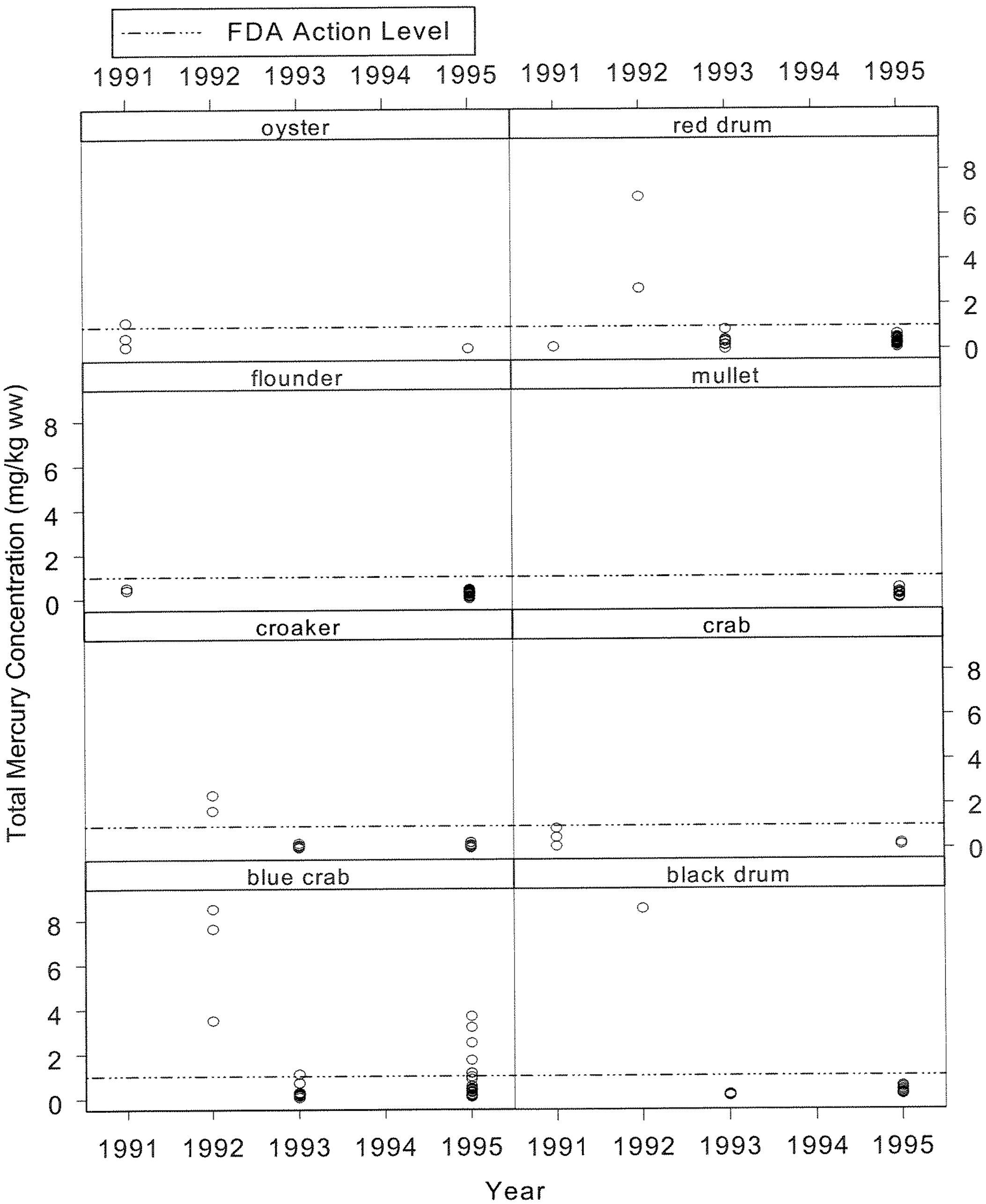

Figure 6.5. Mercury concentrations in fish and shellfish collected from the Purvis Creek/ Turtle River area compared to the FDA action level of 1 mg/kg (ww). Samples in which mercury was not detected are plotted at zero.

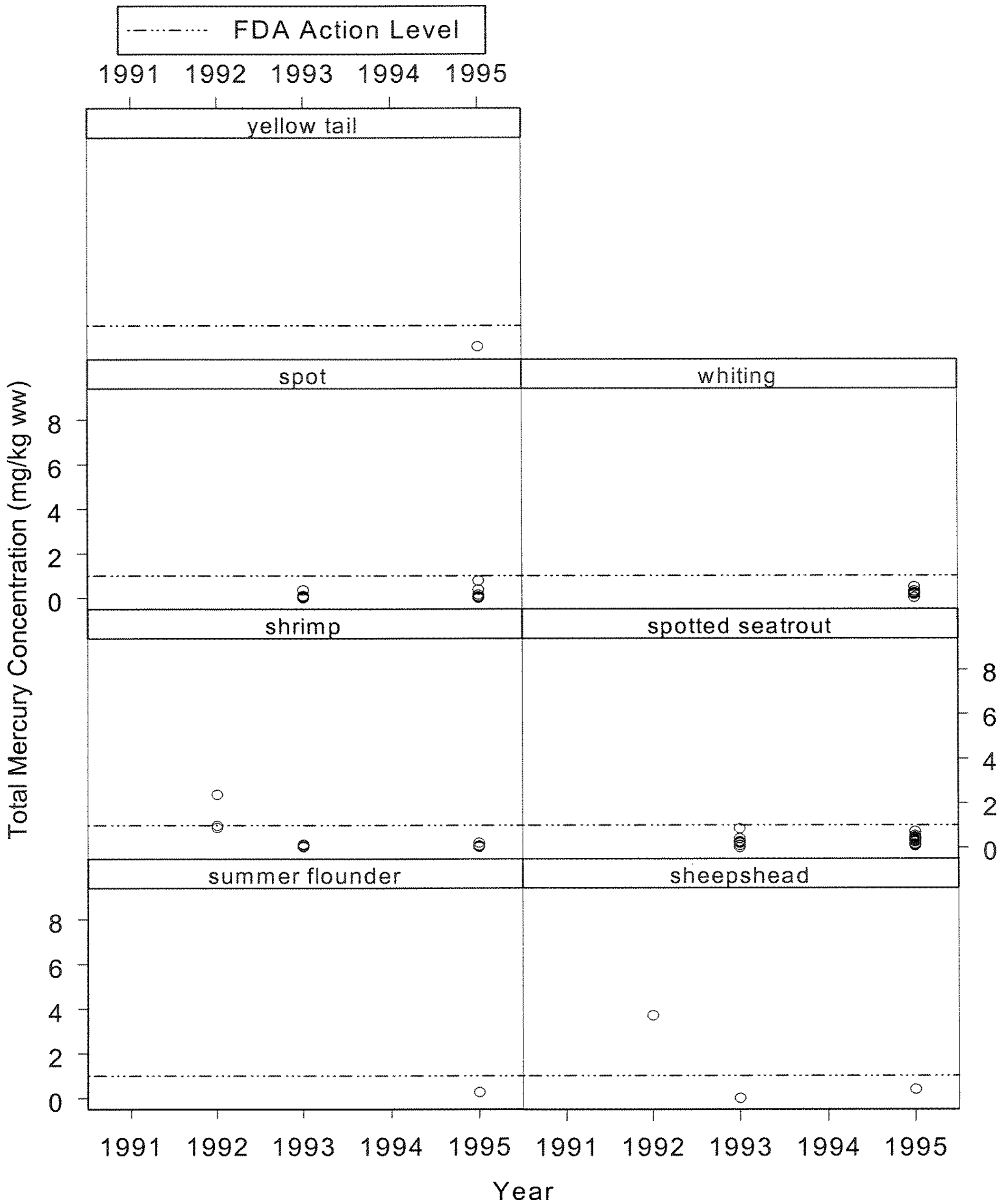

Figure 6.5 (cont.). Mercury concentrations in fish and shellfish collected from the Purvis Creek/Turtle River area compared to the FDA action level of 1 mg/kg (ww). Samples in which mercury was not detected are plotted at zero.

6.4 Conclusions

The concentrations of PCBs and mercury in fish and shellfish in the Purvis Creek and Turtle River area have been sufficient for the State of Georgia to issue advisories and guidelines to limit or ban consumption of fish and shellfish. Consumption advisories or guidelines have been in place since 1992, soon after the first data on PCB and mercury concentrations in fish or shellfish from the area were collected. The State also issued a ban on commercial fishing in the area in 1992 that has remained in place. A comparison of the contaminant data from the site with the State's consumption guideline trigger levels shows that the trigger levels are commonly exceeded.

The FDA tolerance level for PCBs and the FDA action level for mercury are exceeded in the edible portion of some fish and shellfish collected from the Purvis Creek and Turtle River area. The rate of exceedences is greatest for Purvis Creek, where 15% of the samples from 1991 through 1995 exceed the PCB tolerance level and 15% exceed the mercury action level.

Therefore, fish and shellfish in the Purvis Creek and Turtle River area are injured according to the injury definitions contained the DOI NRDA regulations.

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